Disturbance effects of wind farms on birds

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Abstract

Wind energy, like all renewable energy production, is a sustainable resource with a far lower carbon footprint than the burning of fossil fuels and consequently plays a role in mitigating climate change. There has been a rapid rise in its use over the last 15 years. However, there are concerns regarding the potential effects of wind turbines on wildlife, especially birds and bats. Potential effects include disturbance through direct or indirect habitat loss, or fatality through collision with turbine blades.

First, I investigated the potential of using existing bird monitoring data collected by Environmental Impact Assessments (EIAs) and post-construction monitoring for developing an evidence base for the impacts of wind turbines on bird populations. As EIAs are required for wind farms to predict and mitigate potential negative effects, vast amounts of data are gathered by wind farm companies but subsequently not used to assess the effects, due to the associated logistical issues of obtaining these data (despite being officially available in the public domain). Using data from seven wind farm sites, I found that lekking black grouse (*Tetrao tetrix*) that were initially close to the wind turbines (up to 500m) moved locally after wind farm construction but their abundance at the wind farm sites did not change. I conclude that data from EIAs and post-construction monitoring can be used to assess potential effects of wind farms on biodiversity and should be widely available for scientists and policy-makers, ideally via a central data repository.

Second, I investigated whether noise from wind farms might affect birds as bird song could be potentially masked. It is widely known that anthropogenic noise such as traffic and urban noise affects communication in birds. To investigate whether birds are affected by wind turbine noise, I modelled how song from

birds propagates in a hypothetical environment with and without wind turbine noise. I found that songs from species where most of the energy lies in the lower frequencies are masked by wind turbine noise. As this model can predict potential disturbance effects for specific species, it could be used as a tool during EIAs by identifying which species will be affected by wind turbine noise.

Third, as bird communication is masked by wind turbine noise, male-male, male-female and parent-offspring communication could be affected. I investigated if male-male interaction in European robins (*Erithacus rubecula*) is affected by wind turbine noise. Males responded less aggressively during simulated territorial intrusion with background wind turbine noise. This could lead to reduced breeding success as males might not be able to effectively defend their respective territories.

Finally, I explored whether bioaoustic recorders could be used as a replacement for human wildlife surveys. Using the European nightjar (*Caprimulgus europaeus*) as an example, I found that survey methods using bioacoustic recorders are much more accurate and cheaper than surveys by humans. This could aid EIAs as they require wildlife surveys to determine which species might be affected by the development.

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Introduction

Background

In the last 100 years the earth has warmed by 0.74 ^{o}C, with the bulk of the change occurring since 1950. This global warming is most likely to be caused by increases in greenhouse gas (GHG) emissions from human activity (IPCC, 2012). The ecological effects of global warming are already becoming evident: for example, several studies have suggested that spring events are happening earlier for the majority of species (Root et al., 2003; Parmesan and Yohe, 2003). In addition, climate change impacts on food security through extreme weather events such as cyclones, floods and droughts (IPCC, 2007; Schmidhuber and Tubiello, 2007; IPCC, 2014). These extreme weather events also affect water supply and human health (IPCC, 2007, 2014). There is therefore an urgent need to reduce GHG emissions in order to halt the increase in global average temperature.

The Intergovernmental Panel on Climate Change (IPCC) has concluded that emissions will need to start decreasing by 2015 if governments are to meet their Cancun target of limiting the temperature increase to 2 ^{o}C over pre-industrial levels. The urgency of meeting this target has led to a new focus on renewable energy as a way of reducing GHG emissions. Wind power is an important source of renewable energy, providing around 2.1% of electricity worldwide during 2011 (Table 1). This figure may rise to 20% by 2050, according to some projections (IPCC, 2012). The use of on-shore wind farms has increased dramatically over the last decade (GWEC, 2013) (Figure 1). While the exploitation of renewable energy sources will be fundamental to combating climate change, this rapid expansion of wind farm development has raised issues about potential harmful effects on wildlife. Birds are one of the key groups of concern (IPCC, 2012), and

may be affected by wind farms both through direct collision with turbines, and through habitat and ecosystem modifications associated with wind farm developments (Drewitt and Langston, 2006).

Compared to:	Region	% contribution				
		2007	2008	2009	2010	2011
All sources	North America	0.75	1.19	1.69	2.12	2.82
	Central & South America	0.11	0.13	0.20	0.32	0.43
	Europe	2.94	3.34	3.91	4.22	5.11
	Eurasia	0.02	0.03	0.03	0.04	0.08
	Middle East	0.02	0.03	0.03	0.02	0.03
	Africa	0.21	0.22	0.29	0.37	0.42
	Asia & Oceania	0.37	0.54	0.78	1.00	1.36
	World	0.90	1.15	1.45	1.69	2.12
Renewable sources	North America	4.85	7.21	9.59	12.35	14.55
	Central & South America	0.17	0.19	0.29	0.47	0.63
	Europe	13.80	14.81	15.95	16.11	19.47
	Eurasia	0.12	0.16	0.19	0.25	0.50
	Middle East	0.56	1.76	1.81	0.98	1.38
	Africa	1.22	1.31	1.63	2.05	2.41
	Asia & Oceania	2.64	3.53	5.07	6.11	8.54
	World	4.82	5.90	7.13	8.18	10.14

Table 1: Electricity generation by wind power contributing to total electricity generated from all sources and from renewable sources. This table shows the percentage contribution of wind power to total electricity generated (top) and to renewable electricity generated (bottom) for eight international regions for the years 2007 to 2011. Data from U.S. Energy Information Administration (2014).

Current evidence on the effects of wind farms on birds

Wind turbines can affect bird populations in two main ways – directly, via mortality after collision with turbines or towers, or indirectly, the presence or operation of turbines causing disturbance and/or displacement effects. My thesis

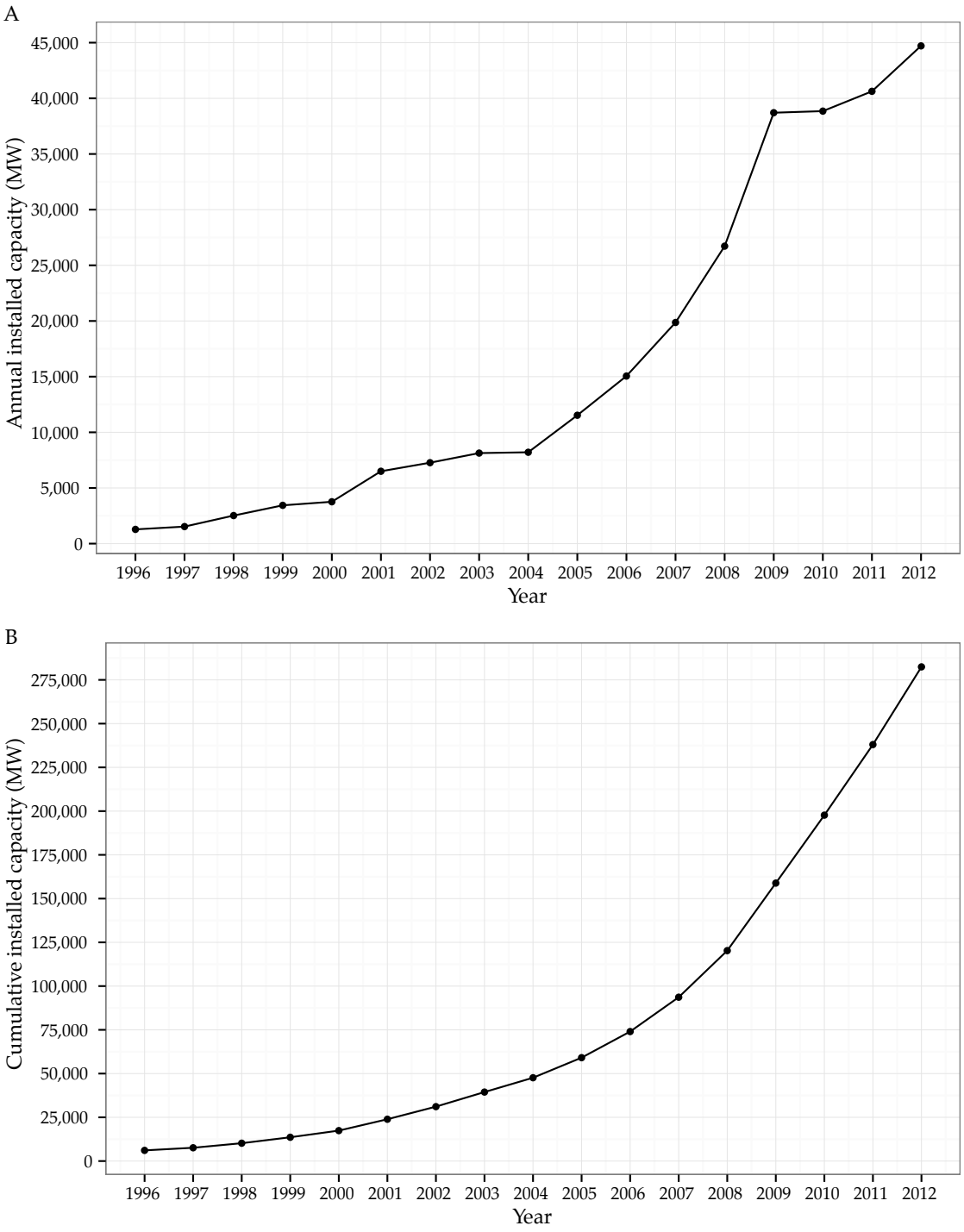

Figure 1: Global installed wind capacity from 1996 to 2012. (A) Global annual installed wind capacity in MW and (B) Cumulative global installed wind capacity in MW. Data from GWEC (2013).

will focus mainly on disturbance effects but first I will briefly focus on collision effects to give a broader description of the impacts of wind farms on birds.

Collision

Every year hundreds of millions of birds are killed due to collisions with a variety of human-made structures, e.g. vehicles, building and windows, power lines, communication towers and wind turbines (Erickson et al., 2001). Bird fatalities due to collisions with wind turbines or associated structures are small compared with other causes of mortality (Erickson et al., 2001). However, the potential effect that mortality caused by collisions with wind turbines may have on certain bird populations should not be underestimated (Hunt, 2002; Madders and Whitfield, 2006). High mortality rates have been reported at some wind farms, for example at the Altamont Pass in California, a large wind farm with over 7,000 turbines, where an estimated 1,100 of raptors are killed each year and thus the wind farm has an estimated mortality rate of 0.15 raptor/turbine/year (Smallwood and Thelander, 2008). At Tarifa in Southern Spain the estimated mortality rate was 0.15 birds/turbine/year for griffon vulture (*Gyps fulvus*) and 0.19 birds/turbine/year for common kestrel (*Falco tinnunculus*) (Barrios and Rodríguez, 2004). It is not only raptors that have been reported to collide with wind turbines as an estimated seven little terns (*Sterna albifrons*), 238 common terns (*Sterna hirundo*) and 84 sandwich terns (*Sterna sandvicensis*) are thought to have collided at a wind farm in Zeebrugge, Belgium (Everaert and Stienen, 2006). By contrast, at a wind farm in Malaga in Southern Spain, only one collision of a common kestrel was recorded during the study period of two years and no collisions were identified for many other species which included raptors, passerines and non-passerines (Farfán et al., 2009). Furthermore, Rothery et al. (2009) reported that possibly two gannets (*Morus bassanus*) had collided at a wind farm at Rothery in England over a post-construction study period of three years and no collisions were reported for eight other seabird spp. The probability of bird collisions depends on factors associated both with the location of the wind farm and the species' flight behaviour. These are discussed in the following sections.

(i) Site-specific factors

The abundance of a species in the wind farm area has repeatedly been identified as one of the major factors affecting bird collision risk (Barrios and Rodríguez, 2004; Carrete et al., 2012). However, collision risk cannot be predicted from abundance alone: some studies found no relationship between species abundance and collision rate (Fernley et al., 2006; Whitfield and Madders, 2006; de Lucas et al., 2008). It is clear that other factors must be involved in modulating collision risk (Orloff and Flannery, 1992, 1996; de Lucas et al., 2008).

One factor that influences avian collision risk is the type of turbine used. For example, lattice towers could provide perches for birds and their attraction could increase collision risk (Orloff and Flannery, 1992; Smallwood and Thelander, 2004; Percival, 2005), but other studies have not supported this theory (Barrios and Rodríguez, 2004; de Lucas et al., 2008). Turbines may also differ in height, for example at Altamont Pass the hub height differed from 12m for the smallest turbine to 46m for the tallest turbine. However, this was not reported as a factor that correlates with collision risk (Orloff and Flannery, 1992, 1996) and in an independent meta-analysis Hötker et al. (2006) found only a weak relationship.

Another factor influencing avian collision risk is the number of turbines and their layout (Langston and Pullan, 2003). A wind farm consisting of a large number of turbines (e.g. over 7000 turbines at Altamont Pass) may be associated with a large number of fatalities overall even if the collision risk per turbine is low (Percival, 2003; Langston and Pullan, 2003). In addition, a wind farm where turbines are positioned close to one another allows less space for birds to successfully manoeuvre between them (Hunt, 2002; Percival, 2005). Furthermore turbines located at the end of a row were reported to have higher collision rates at Altamont Pass (Orloff and Flannery, 1992, 1996; Smallwood and Thelander, 2004), although not at Tarifa (de Lucas et al., 2008) but the underlying causes for this difference are unknown.

Topographical features have also been suggested to influence collision risk in birds. For example, to successfully evade turbines at higher altitudes vultures

require more lift, which might not always be available (de Lucas et al., 2008). Vultures need more lift to fly over, and therefore successfully evade, turbines at higher altitudes (de Lucas et al., 2008). Furthermore, many raptors use updrafts to aid their flight and thus areas with weaker updrafts can have higher mortality rates. For example, at a site in Spain vulture mortality per turbine was higher in areas with gentle slopes producing weaker updrafts (de Lucas et al., 2008). In addition, wind farms located near features such as a sharp change in relief (e.g. plateau edges) and/or on mountain ridges caused higher raptor mortality (Percival, 2005; Hötker et al., 2006). Hunt (2002) found that slope was not related to the number of fatalities but this might be caused by the fact that other factors (such as turbine spacing) were more important in the situation studied.

Finally, some studies showed correlations between prey availability and golden eagle (*Aquila chrysaetos*) collision mortality (Hunt, 2002; Smallwood and Thelander, 2004), perhaps because hunting raptors might not see the turbines as they keep their eyes on their potential prey.

(ii) Species-specific factors

In addition to the site-specific factors discussed above, collision risk can also be affected by interspecific variation in behaviour and physiology. Birds in flight tend to focus their attention on what is below them to allow for effective detection of foraging opportunities, but they also focus on conspecifics or roost sites (Martin, 2011). They may have learnt to expect the open airspace above vegetation to be highly predictable and usually free of hazards (Martin, 2011). In addition, the visual system of birds provides high resolution vision in the lateral fields but not in the frontal field. As a result, birds (particularly in flight) may have limited awareness of what is in front of them (Martin, 2011), increasing collision risk with 'unexpected' objects such as wind turbines.

Variation in flight manoeuvrability, which depends largely on morphology (Drewitt and Langston, 2008) is another factor affecting collision risk. Larger, relatively heavier species tend to have a lower flight manoeuvrability and are thus less able to avoid wind turbines when necessary (Garthe and Hüppop,

2004). Many soaring birds are less manoeuvrable as they have a weak-powered flight and use updrafts or thermals to power their flight (Tucker, 1971; Pennycuick, 1975; de Lucas et al., 2008). The number of flights, their duration and height also influence collision risk (Garthe and Hüppop, 2004; Drewitt and Langston, 2008). For example, many passerines making local movements, as opposed to during migration, tend to fly lower than the rotor swept area of larger turbines reducing the risk of collision (Hötker et al., 2006).

(iii) Other factors

A number of additional factors affecting bird collision risk that are not directly related either to the properties of the wind farm or bird biology and ecology have been identified. Certain weather conditions can influence flight ability. For example, heavy winds will affect flight manoeuvrability (Langston and Pullan, 2003). Furthermore, fog and heavy rain will impede vision and thus affect collision risk (Larsen and Guillemette, 2007). It has been shown that collision risk changes with different seasons. In winter, lower temperatures mean that thermal updrafts are less common, affecting the flight ability of soaring birds. Indeed, de Lucas et al. (2008) found higher collision rates during winter than other seasons. Another study concerning little, common and sandwich terns showed that collision risk was higher during chick provisioning (Everaert and Stienen, 2006).

Disturbance

Substantial amounts of infrastructure (e.g. access tracks) are created during construction of wind farms. This, combined with the 'footprint' of the turbines themselves, causes a certain amount of direct habitat loss and/or fragmentation. While this loss of habitat is negligible for smaller wind farms, when the development consists of hundreds or even thousands of turbines this loss can be considerable. Furthermore, birds may also avoid the area surrounding the wind farm, causing indirect habitat loss. This has been reported for different seasons and for different species, primarily raptors, geese, ducks and waders (Hötker et al., 2006) (Table 2).

Species	Scientific name	Country	Disturbance	Season	Reference
American Kestrel	*Falco sparverius*	US	Yes	summer	Garvin et al. 2011
Bewick's Swan	*Cygnus bewickii*	Netherlands	Yes	winter	Fijn et al. 2007
Black Grouse	*Tetrao tetrix*	Austria	Yes	breeding	Zeiler and Grünschachner-Berger 2009
Common Eider	*Somateria mollissima*	Denmark	Yes	winter	Larsen and Guillemette 2007
Cormorant	*Phalacrocorax carbo*	UK	Yes	breeding	Rothery et al. 2009
Corvids	*Corvidae*	UK	No	winter	Devereux et al. 2008
Dunlin	*Caldris alpina*	Norway	Yes	breeding	Bevanger et al. 2010
Eurasian skylark	*Alauda arvensis*	UK	No	winter	Devereux et al. 2008
Gamebirds		UK	No	winter	Devereux et al. 2008
Golden Plover	*Pluvialis apricaria*	UK	Yes		Pearce-Higgins et al. 2009
Golden Plover	*Pluvialis apricaria*	Norway	Yes	breeding	Bevanger et al. 2010
Golden Plover	*Pluvialis apricaria*	UK	No	breeding	Douglas et al. 2011
Granivores		UK	No	winter	Devereux et al. 2008
Great Black-backed Gull	*Larus marinus*	UK	No	breeding	Rothery et al. 2009
Northern Harrier	*Circus cyaneus*	US	Yes	summer	Garvin et al. 2011

Table 2: List of examples of wind farm disturbance by certain species. This list was constructed via a literature search. (Continued on next page)

Species	Scientific name	Country	Disturbance	Season	Reference
Pheasant	*Phasianus colchicus*	UK	Yes	winter	Devereux et al. 2008
Pink-footed Goose	*Anser brachyrhynchus*	Denmark	Yes		Larsen and Madsen 2000; Madsen and Boertmann 2008
Red-tailed Hawk	*Buteo jamaicensis*	US	Yes	summer	Garvin et al. 2011
Sandwich Tern	*Sterna sandvicensis*	UK	No	breeding	Rothery et al. 2009
Tundra Bean Goose	*Anser serrirostris*	Netherlands	Yes	winter	Fijn et al. 2007
Turkey Vulture	*Cathartes aura*	US	Yes	summer	Garvin et al. 2011
Wheatear	*Oenanthe oenanthe*	Norway	Yes	breeding	Bevanger et al. 2010
White tailed eagle	*Haliaeetus albicilla*	Norway	Yes	breeding	Bevanger et al. 2010; Dahl et al. 2012
Willow Ptarmigan	*Lagopus lagopus*	Norway	No	breeding	Bevanger et al. 2010
Willow Ptarmigan	*Lagopus lagopus scotica*	UK	No	breeding	Douglas et al. 2011

Table 2: List of examples of avoidance behaviour of certain species. This list was constructed via a literature search.

However, birds do not always avoid turbine sites. For example, no disturbance effects have been found for most passerine species (Devereux et al., 2008; Farfán et al., 2009), or a range of other species e.g. willow ptarmigan (*Lagopus lagopus*) (Bevanger et al., 2010; Douglas et al., 2011). Thus, avoidance behaviour appears to be species-specific, and it is unclear why certain species are affected whilst others are not. While we do not understand all the mechanisms driving avoidance behaviour, a range of contributing factors have been identified which I will outline below.

Firstly, noise produced by turbines could affect bird communication or foraging efficiency and birds might therefore perceive areas close to wind farms as of lower habitat quality. Most noise studies on animals have focused on the effects of urban or traffic noise. For example, great tits (*Parus major*) adjust the pitch of their song in response to urban noise (Slabbekoorn and Peet, 2003) and traffic noise is correlated with a reduction in reproductive performance (Reijnen et al., 1996; Halfwerk et al., 2011). In another study, noise lowered chaffinches' (*Fringilla coelebs*) foraging efficiency (Quinn et al., 2006). There is limited information currently published on the impacts of wind turbine noise. One study showed that the antipredator behaviour of ground squirrels (*Spermophilus beecheyi*) is affected by wind turbine noise (Rabin et al., 2006), but whether such effects can be generalized to other species is currently unclear. Zeiler and Grünschachner-Berger (2009) suggested that black grouse (*Tetrao tetrix*) may have left a wind farm site because of song disruption. However, the impacts of wind farm noise on bird distribution has not been directly addressed and that is a major part of this thesis.

Secondly, increased human activity associated with wind farms could affect bird populations (Langston and Pullan, 2003; Madders and Whitfield, 2006; Zeiler and Grünschachner-Berger, 2009). Such an increase would most likely be due to wind farm maintenance, but could also result from increases in tourism. After the construction of a wind farm in Norway, hiker activity increased as access to the area was improved through the newly created tracks that accompanied the wind farm development (Bevanger et al., 2010). Human disturbance is known to affect bird distribution in a number of ways including fleeing to an approaching person

and thus reducing intake rates (de Boer and Longamane, 1996; Goss-Custard et al., 2006) and increasing predation rates of nesting birds (when they flee their nests) (Lord et al., 2001).

Third, physical properties of the wind farm such as turbine size and layout may alter bird distributions. Larger turbines can have a greater effect on resting birds than smaller turbines (Hötker et al., 2006; Madsen and Boertmann, 2008), which could be because larger turbines are more spaced out and thus cover a larger area. In contrast, breeding birds, particularly songbirds, were less affected by larger turbines (Hötker et al., 2006). Moreover, within wind farms, turbines can be positioned in clusters or in rows. One study suggested that clusters might lead to a greater disturbance of pink-footed geese (*Anser brachyrhynchus*), as a cluster layout often coincides with their preferred habitat of open landscape (Larsen and Madsen, 2000).

Finally, the construction of the wind farm might cause more of an effect than the operational state (Douglas et al., 2011; Pearce-Higgins et al., 2012). If this is the case it would be expected that the birds return to the site over time after construction is finished. This has only been reported for a few cases (e.g. Madsen and Boertmann, 2008) and some studies have reported that there is no habituation (Hötker et al., 2006; Stewart et al., 2007; de Lucas et al., 2008), but further longer-term studies are necessary to test this hypothesis.

In addition to indirect habitat loss, avoidance may lead to habitat fragmentation - the turbines lowering habitat quality in the surrounding area and thus breaking up a single patch of habitat into several smaller high quality patches. Further studies are needed to fully understand the disturbance effects of wind farms on birds. In particular, raptors are a key group that have been shown to be at risk of collision but it is challenging to study the population-level impacts of turbines due to the low breeding densities of many species of raptor (Newton, 1979). It is worth adding one final note of caution: some of the effects of turbines on birds might have gone unnoticed as studies might not have been long enough to detect an effect (Garvin et al., 2011) or due to a lack of Before-After Control-Impact (BACI) studies (Madders and Whitfield, 2006). The before-after design involves

data collection prior to construction and compares it with data after construction. Collecting data before and after construction from the wind farm site and a control site is known as a BACI design.

Population-level effects

Both direct collision mortality and disturbance effects may have population-level consequences. These effects are likely to be highly species-specific and I discuss the potential impacts at a population-level below.

Consequences of direct collision mortality

In contrast to disturbance effects, population-level consequences of collision mortality are thought to be more direct. Mortality from collisions could have a major impact on the population level of a species (Langston and Pullan, 2003), particularly for long-lived species with low productivity (Langston and Pullan, 2003; Hötker et al., 2006). Species with a small global range or population size might be particularly vulnerable. It is therefore important to look at the status of the birds that are using the proposed wind farm site to determine the potential effects. Population effects may not be immediately apparent. For example, the area could have become an ecological sink where new adults are constantly recruited from areas outside the wind farm (Smallwood and Thelander, 2008).

Consequences of disturbance effects

The population-level consequences of disturbance effects are difficult to quantify and few studies have done so (Pearce-Higgins et al., 2012). Habitat loss caused by turbines is expected to cause a decrease in the overall quality of remaining habitat (Larsen and Madsen, 2000; Madders and Whitfield, 2006). The population-level response to this decrease in habitat quality depends on whether alternative habitat is available (Langston and Pullan, 2003). For example, geese and swans moved from the control areas to the wind farm area only when food availability in the control area was depleted (Fijn et al., 2007). Furthermore, birds might be displaced into less suitable habitat because optimal habitat might already support

the maximum number of that species (e.g. insufficient nesting locations available or a shortage of food resources in the area), which may reduce their ability to survive and reproduce (Madders and Whitfield, 2006; Dahl et al., 2012). This drop in productivity affects long-lived species with low annual productivity and slow maturation more than short-lived species with higher annual productivity (Langston and Pullan, 2003; Hötker et al., 2006).

Alternatively, avoidance of turbine development areas may cause flights (e.g. between breeding and foraging grounds or migration flights) to be altered: the so-called barrier effect. Changes in flight paths may incur extra energy costs as travelling distances are increased. Increased energy costs could adversely affect survival or breeding success. For example, while flight lines of breeding little terns, common terns and sandwich terns feeding young passed through a wind farm area, the same site was avoided during the non-breeding season, suggesting that they could not afford the extra flight time during the breeding season (Everaert and Stienen, 2006). Migrating common eiders (*Somateria mollissima*) and geese have been reported to fly around an offshore wind farm in Denmark (Desholm and Kahlert, 2005), and population-level consequences of the barrier effect for migratory populations are unclear, although expected to be limited if increases in flight time are relatively small (Desholm, 2003).

Placing my thesis chapters in context

My four data chapters have different backgrounds and below I outline the background to each data chapter and how it fits within the broader question of impacts of wind farms on bird populations.

Environmental Impact Assessments

In Europe, Competent Authorities require Environmental Impact Assessments (EIAs) to be carried out prior to any wind farm developments taking place. These aim to ensure that the development is placed in a suitable location which minimises adverse impacts on wildlife (Directive 2011/92/EU, 2012). In brief, EIAs require a range of ecological surveys to be carried out, including those to

determine which bird populations might be affected by any development. In addition, the sensitivity of those populations to any impact is determined and the scale of any potential effects is assessed. Finally, recommendations are made as to the acceptability of the predicted effects of the proposed development (Percival, 2003). Outside Europe, there is little information on requirements that is easily accessible. Canada (Kingsley and Whittam, 2005) and Mexico (Martínez, 2008) have similar guidelines in place to those in Europe. In the United States, survey requirements vary extensively by state; some states have very detailed guidelines about citing wind farms while others do not have any (Jodi Stemler Consulting, 2007).

Effects of wind farms on bird populations are only possible to measure post-construction. There are some striking examples of significant impacts of wind farms on birds, as in the cases of Altamont and Tarifa (for details see above) (Barrios and Rodríguez, 2004; Smallwood and Thelander, 2004). Although no EIAs were performed before Altamont and Tarifa were constructed, it is important to recognise that not all effects can be successfully predicted (Ferrer et al., 2012), at least with our current level of knowledge. For example, some of the highest mortality rates have been reported at sites where collision risk was estimated to be sufficiently low during risk assessment studies conducted before construction (Ferrer et al., 2012). Alternative mitigation measures are required in such cases. For example, at Altamont, repowering wind farms, by replacing old turbine with modern turbines, is taking place to reduce bird mortality (Smallwood and Thelander, 2004). Repowering wind farms can reduce bird mortality by avoiding areas which are known to have high mortality rates. Smallwood et al. (2009) suggested repowering could reduce 70% of the mortality caused by the wind farm. In addition to repowering, turbines could be stopped at times when collision risk is highest. For example, in a recent case study, mortality was halved, while losing only 0.07% of energy production (de Lucas et al., 2012). Finally, those turbines that cause the majority of fatalities could be removed (Sterner, 2002). As not all effects can be successfully predicted, many wind farm projects require post-construction monitoring. Given how widely EIAs and post-construction monitoring are undertaken, there is a substantial

amount of ecological data generated that are only used to address specific questions about a single development and are rarely used to address other questions, although there are some exceptions which use data from multiple sites (e.g. Ferrer et al., 2012). In my first data chapter, I explore the use of data collected during EIAs and post-construction monitoring to study the possible impacts of wind farms on birds. I will focus on black grouse, one species at potential risk from wind turbines (European Commission, 2011).

Noise and acoustic communication

As mentioned previously, turbine noise might cause disturbance and/or avoidance of the wind farm area. This is due to the possibility of wind farm noise affecting acoustic communication in animals. Many species use acoustic signals to communicate with each other. Birds, in particular, use sound signals which contain information about the sender of the signal. This information may include the sender's species, sex, or territorial and reproductive status, but could also include behavioural information. For example, it may contain the probability of responding aggressively or sexually to a potential receiver (Emlen, 1972). According to the acoustic adaptation hypothesis, signal evolution depends on both the habitat and the range over which the signal must be broadcast (Morton, 1975). For a receiver to be able to detect this signal, it will need to be louder than the background noise in that habitat. Increased noise levels will reduce the distance over which a signal can be detected by increasing the hearing threshold of the receiver. This masking effect of noise can have severe consequences. Several studies have shown reduced breeding densities and/or success near noisy roads (Reijnen et al., 1996; Rheindt, 2003; Parris and Schneider, 2009; Halfwerk et al., 2011). However, it is unclear whether confounding variables which are present at any road, such as collisions and edge effects, might have been the cause of these patterns. Recently a study by McClure et al. (2013) controlled for these variables by introducing traffic noise in a roadless environment and found reduced bird densities at these sites, suggesting that traffic noise is the cause of these reduced densities.

Furthermore, to overcome the masking effect of noise, senders might change

their signal. Several bird species have been found to increase their minimum frequency in response to increased noise levels. These include: European robins (*Erithacus rubecula*), nightingales (*Luscinia megarhynchos*), blackbirds (*Turdus merula*), great tits and common chiffchaffs (*Phylloscopus collybita*) (Slabbekoorn and Peet, 2003; Brumm, 2004b; Nemeth and Brumm, 2009; Verzijden et al., 2010; McLaughlin and Kunc, 2013). Interestingly, a recent study by Halfwerk and Slabbekoorn (2009) showed that when great tits were exposed to noise with most of its acoustic energy in the higher frequencies they reduced their minimum frequency and when exposed to noise with its acoustic energy in the lower frequencies (like urban and traffic noise) they increased their minimum frequency. These results suggest that bird vocal responses are different depending on the acoustic signature of anthropogenic noise. Nemeth et al. (2013) recently suggested that the increase in minimum frequency might be due to vocal constraints. They found that blackbirds increase their amplitude as well as increasing the minimum frequency of their song. Higher frequencies are easier to sing at higher amplitudes and it may not be physically possible for birds to sing lower frequencies at higher amplitudes and therefore they switch to a higher frequency. However, not every species is likely to have the vocal plasticity to increase the frequencies and/or increase the amplitude of their song. These species might avoid noisy areas altogether. Furthermore, several recent studies have found that an increase in minimum frequency is not as beneficial as an increase in amplitude. In great tits an increase of 200 Hz in frequency only increased the maximum communication distance by 10%, but an increase of 5 dB in amplitude increased the maximum communication distance by 50% (Nemeth and Brumm, 2010).

It is currently unknown what effect an increase of amplitude and/or increase of frequency might have on a population. Lower frequency signals have been found to signal aggression (Morton, 1977). For example, canyon wrens (*Catherpes mexicanus*) lowered their minimum frequencies in response to a simulated intrusion through playback (Benedict et al., 2012). These signals may be used to win contests without physical conflict as the intruder could be informed of the superior fighting ability and motivation to attack of the sender (Maynard Smith

and Price, 1973). Increasing the frequency of the song might therefore be interpreted as less threatening and may result in failure to deter a rival male, leading to physical conflict and possible reduced reproductive success.

Most research has focused on the effects of urban and traffic noise. However, with the increasing number of wind farms, the effects of turbine noise should be considered, especially since wind farms are usually built away from urban areas. The acoustic signature of wind turbine noise has most of its energy in the lower frequencies (Oerlemans et al., 2007). Therefore, we could expect similar effects as to what we observe with urban and traffic noise but interestingly wind turbines are often situated in remote locations that mean their impacts could be relevant to a range of species that are not subject to urban or frequent traffic noise. I investigate the effects of wind farm noise on acoustic communication in birds in two different chapters. In my second chapter, I describe a general model for predicting which frequencies and which species could be affected by wind turbine noise. Specifically, I model the area where a signal could be heard and interpreted (active space) in quiet and noisy conditions and calculate the amount of reduction in active space. In my third chapter, I investigate the effects of wind turbine noise on male-male interaction in European robins. Most research has focused on the differences in song characteristics between urban and forest environments either by recording songs in these two habitats or experimentally increasing noise levels in a quiet environment. However, effects on interactions between for example male-male, female-male, parent-offspring remain unknown. Only recently has a study investigated the effects of urban noise on male-female interaction in great tits and showed that urban conditions may undermine the selective advantage of using low frequency song types, as high frequency songs were preferred by females in noisy conditions (Halfwerk et al., 2011). In addition, they found that the usage of low frequency songs reduced extra-pair copulations and therefore is related to reproductive success (Halfwerk et al., 2011). As great tit males sing high frequency song types in noisy conditions, it might be more difficult for high quality males to distinguish themselves from competitors in noisy environments, which are identified by low frequency song types. It is important to investigate if there are changes in these behaviours as this will give us more understanding of

the effect that noise has and whether an increase in amplitude or frequency has consequences for the signaller.

Bioacoustic recorders

In my final chapter I will investigate whether bioacoustic recorders can be used as a replacement for human wildlife surveys. To be able to monitor and protect endangered species we need accurate information on their numbers and where they live. The traditional method to collect this information is via human surveys, where a surveyor goes out in the field and counts the number of individuals in a particular area. In the UK this is typically via breeding bird surveys (e.g. via common bird census (Sutherland et al., 2004a) or following methods described in Gilbert et al. (1998) or for moorland breeding birds following Brown and Shepherd methodology (Brown and Shepherd, 1993)). Breeding bird surveys are widely used in EIAs, but have several disadvantages. Firstly, the behaviour or ecology of the species of interest could reduce detectability. For example, nocturnal species are only active at night and therefore would require night surveys when visibility is greatly reduced and the surveyor would mostly rely on hearing to detect the species (which is very reliant on the frequency of calling). In addition, the species might be well camouflaged or very secretive, reducing its detectability. Secondly, because of the behaviour or ecology of the species of interest, expert surveyors might be needed. Thirdly, these human surveys are expensive to carry out. Automated recording via bioacoustic recorders might offer a cheaper alternative. I investigated this surveying method by comparing it to the traditional method using the European nightar (*Caprimulgus europaeus*) as an example. Nightjars are nocturnal and well camouflaged and therefore are not easy to detect during traditional surveys. They are also in decline in the UK and are listed on the Annex I of the Birds Directive. This means that the governments of European Union countries are required to protect the habitat of this species. In the UK, Special Protection Areas (SPAs) are designated for nightjars when 1% or more of its national population lives within a specified location. Furthermore, if bioacoustic recorders are a better survey method than traditional methods, it would contain more accurate

information on species abundance and their distribution. This in turn would aid EIAs and post-construction monitoring for wind farms and other developments as more accurate predictions of the effects of the development can be made.

Overall thesis plan and my contribution to each data chapter

Overall my thesis aims to: (i) demonstrate the potential use of existing bird monitoring data collected by EIAs and post-construction monitoring for developing an evidence base for the impacts of wind turbines on bird populations (Chapter 1); (ii) advance our understanding of the impacts of wind turbine noise on birds (Chapters 2 and 3) and (iii) to explore a potential new bird monitoring method using audio recorders (Chapter 4).

My data chapters are written as papers for publication. I accessed, analysed and wrote Chapter 1 (which has been submitted for publication). I collected the data, analysed and wrote Chapters 2 and 3 (the data collection for Chapter 3 was done in collaboration with Jonathon Dunn) and intend to submit them as papers for publication shortly. I collected the data for Chapter 4 with the help of four fieldworkers and I analysed the data and wrote the paper which is published in PLoS ONE.

Chapter 1. Using Environmental Impact Assessments and post-construction monitoring data to develop an ecological evidence base: the impacts of wind farms on black grouse in Scotland

1.1 Abstract

Ecological data are routinely collected for Environmental Impact Assessments (EIAs) and post-consent planning requirements to assess potential negative impacts of developments on wildlife. Such data are almost always obtained from a single site and this often prohibits robust statistical analysis due to insufficient replication. Here, we integrate data collected during EIAs and post-construction monitoring from multiple sites to study the impact of wind farm developments on the distribution and abundance of black grouse. We show that the construction of wind turbines at these seven sites had no detectable effect on the abundance of lekking black grouse, but that leks within 500m of the nearest planned wind turbine moved locally after construction. This effect was not observed for leks greater than 500m from a wind turbine. There are several reasons which, individually or in combination, could underlie the localised movement of black grouse we report. These include the operation of wind farm infrastructure, volume of visitors, changes in land management both within and surrounding the site, and habitat enhancement measures designed to attract black grouse to specific areas away from the wind farm. We demonstrate that ecological data routinely collected by EIAs and post-construction surveys from multiple projects can be combined to provide a robust ecological evidence base on which to inform development decisions. We recommend that easily-accessible data repositories be maintained by regulatory authorities to enable the development of a robust ecological evidence base to guide planning decisions across a wide range of different wildlife.

1.2 Introduction

Change in land-use has had a significant impact on biodiversity and continues to do so (Sala et al., 2000; Foley et al., 2005). Consequently, there is a need to synthesise the results from robust ecological studies in order to understand and predict the consequences of this change. Where the change is due to a development, the standard approach is to conduct an Environmental Impact Assessment (EIA), which is intended to minimise and, where necessary, mitigate the potential negative impacts of the development. EIAs are carried out globally and are subject to different regulations dependent on their location. For example, in the European Union (EU) the process was originally outlined in EU Directive 85/337/EEC (1985) on Environmental Impact Assessments in 1985 and subsequently amended (Directive 2011/92/EU, 2012). One recent estimate suggested that there are around 16,000 EIAs carried out across the EU each year (GHK, 2010), with follow on post-construction monitoring required, under planning regulations, on a subset of these. Given how widely EIAs and post-construction monitoring are undertaken, there is a substantial amount of ecological data generated that are only used to address specific questions about a single development and are rarely used to address other questions, although there are some exceptions which use data from multiple sites (e.g. Ferrer et al., 2012).

The specific purpose for which EIA and post-construction monitoring data are gathered and the associated logistical issues of obtaining this data (despite being officially available in the public domain) mean that the data are rarely marshalled to help address significant questions at larger spatial and temporal scales. This is unfortunate given the difficulties in determining cumulative impacts of development on landscapes (e.g. Masden et al., 2010) and the uncertainty of the effect of EIAs on biodiversity conservation (Sutherland et al., 2009). Furthermore, the demand by both scientists (e.g. Sutherland et al., 2004b) and policy makers (e.g. Parliamentary Office of Science and Technology, 2011) for management and policy to be based on strong scientific evidence suggests that new ways of making data available for analysis would be very welcome. We

explore the potential of data gathered during the EIA process and post-construction monitoring to answer such larger scale questions by asking what impact wind farm development has on the distribution and abundance of a species of conservation concern.

Based on data obtained from the Netherlands, the number of EIAs carried out due to wind farm development has increased significantly from 0.6% of all EIAs in 2006-2008 to 5.9% of the EIAs in 2013 (www.commissiemer.nl, 2013). However, an estimation across Europe could not be made as there is no specific information on this. Wind energy, like all renewable energy production, is a sustainable resource with a far lower carbon footprint than the burning of fossil fuels and consequently plays a role in mitigating climate change (IPCC, 2012). There has been a rapid rise in its use over the last 15 years (IEA, 2010; GWEC, 2013). However, there are concerns on the potential effects of wind turbines on wildlife, especially birds and bats. Potential effects include disturbance through direct or indirect habitat loss (e.g. Pearce-Higgins et al., 2009; Dahl et al., 2012), or fatality through collision with turbine blades (e.g. Orloff and Flannery, 1992; Horn et al., 2008; Smallwood and Thelander, 2008; de Lucas et al., 2012). Much focus has been on birds, with larger, less manoeuvrable, species tending to be at greatest risk; these include raptors, geese and gamebirds (Barrios and Rodríguez, 2004; Hötker et al., 2006; Fijn et al., 2007; Smallwood and Thelander, 2008; Zeiler and Grünschachner-Berger, 2009; Garvin et al., 2011; Dahl et al., 2012). A proper understanding of the risks posed by wind farms is important for at least two reasons: conservationists are concerned about the potential effects on vulnerable species, and at the same time planning applications from wind energy companies can be refused or subject to costly delays if wildlife is judged to be threatened. Potential effects like disturbance and collision could be substantially reduced by careful placement of wind farms and their turbines, in particular by building them away from known breeding sites of vulnerable species. Therefore, during the planning phase of a wind farm development, EIAs are carried out to advise about its potential effects. Ideally these decisions would be based on the widest possible evidence base, including survey data collected in a consistent manner from a range of sites over a meaningful time period, in order to draw

robust conclusions. However, the expense involved means that this kind of dataset is rarely, if ever, available and planning decisions are being made in its absence, and are typically based on collective experience or professional opinion (Hill and Arnold, 2012).

In this paper, we explore the usefulness of data collected during EIAs and post-construction monitoring to study the possible impacts of wind farms on birds. We focus on black grouse, *Tetrao tetrix*, a species considered at risk from wind turbines (European Commission, 2011). Although little research has been conducted on the impact of wind farms on black grouse, the one study that has been performed at a wind farm in the Austrian Alps showed that the abundance of black grouse at the wind farm site decreased rapidly after construction (Zeiler and Grünschachner-Berger, 2009). However, this decline also coincided with continued shooting of black grouse at the site and the construction of a major ski-lift operation with an associated large increase in human disturbance. Thus, at this site the cause of the black grouse decline is far from clear. Here we aim to provide an evidence base of population changes across seven sites where wind farms have been constructed. Our data provides insight into both the abundance and distribution of black grouse from before and after the wind farms were constructed.

We address simple, yet fundamental, questions with our data set: is wind farm construction associated with a change in the (i) numbers and (ii) the distribution of lekking black grouse in the surrounding area? The interpretation of our results is complicated by background declines of British black grouse populations, habitat enhancement aimed at benefitting black grouse at some of these wind farm sites (as mitigation for perceived impacts) and other changes in habitat (both within and adjacent to each site). We discuss our results in light of these factors and insights into the use of EIA and post-construction monitoring data to answer ecological questions.

1.3 Material and methods

1.3.1 Gathering of data

We contacted six different wind farm developers active in Scotland to ask for data from wind farm sites where black grouse occurs. From these only ScottishPower Renewables responded with data. In addition, we obtained data from one site through the Central Scotland Black Grouse Study Group.

1.3.2 Wind farm sites

Seven sites in southern Scotland were surveyed for lekking black grouse during the breeding season (April-May) before and after construction of the wind farms. These data are covered by restrictions and so the site names are not given here, but are referred to by numeric identifiers. We are unable to provide information on the number of turbines or the year that each site became operational, although the earliest became operational in 1995 and the latest in 2010. Six of the study sites incorporated Habitat Management Plans (HMPs) as mitigation for potential impacts of wind farm development on black grouse. This is because wind turbines in the UK are currently perceived as having a negative impact on the species, largely based on the Precautionary Principle (UNESCO COMEST, 2005). The habitat modifications on the sites consisted of tree planting (sites 1, 3, 4, 5 and 7), predator control (culling of foxes and crows) (site 1 and 7), grazing restrictions (site 1 and 7), removal of redundant fences (site 7), marking of existing fences (site 7), tree felling (site 3, 6 and 7), creating small wet areas (site 3 and 7) and blanket bog restoration (site 3 and 6) in addition to the construction of the turbines (see Table 3). These modifications took place in the wind farm area and/or areas immediately adjacent to, or within 400m of, the area containing the wind turbines. The areas of modifications varied greatly between sites but at five of the six sites these extended at least several kilometer squares in size. The majority of this enhancement work took place at the end of the construction period. However, at sites 1, 3, 6 and 7 the habitat modifications are still on-going as the habitat is naturally changing as a result of felling and grazing restrictions. In addition, at site 6 blanket bog is being restored by ground treatments. These

habitat changes are representative of the management and natural changes that take place throughout the uplands. Only at site 2 there was no HMP to mitigate for potential effects of wind farm development on black grouse.

Habitat modification	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Tree planting	x		x	x	x		x
Predator control	x						x
Grazing restrictions	x						x
Removal or marking of existing fences							x
Tree felling			x			x	x
Creating small wet areas			x				x
Blanket bog restoration			x			x	

Table 3: Overview of the habitat modifications at each site. The habitat modifications on the sites consisted of tree planting, predator control (culling of foxes and crows), grazing restrictions, removal of redundant fences, marking of existing fences, tree felling, creating small wet areas and blanket bog restoration in addition to the construction of the turbines. These modifications took place in the wind farm area and/or areas immediately adjacent to, or within 400m of, the area containing the wind turbines. The areas of modifications varied greatly between sites but at five of the six sites these extended at least several kilometer squares in size. The majority of this enhancement work took place at the end of the construction period.

1.3.3 Data collection

Data on black grouse distribution and abundance were obtained from the Central Scotland Black Grouse Study Group and ScottishPower Renewables. Data collection at sites 1, 3, 4 and 5 was shared between five different ecological consultancy companies and the RSPB in different years (e.g. site 3 was surveyed by company A in survey years 1 and 4 and company B in survey years 8 and 9, by company C in survey year 10 and 12 and finally by RSPB in survey years

13-15). Data collection at site 2 and 6 was carried out by two different ecological companies (a different company was used for each site) and the data from site 7 was collected by the RSPB. The sites were surveyed both before and after construction of the wind farm for any displaying black grouse (see Table 4). Survey methods followed the methods described in Gilbert et al. (1998). In short, a number of visits, varying from one to six, were undertaken to count the total number of birds attending suitable areas for lekking. It is possible that the number of visits altered before and after turbine construction: to test for this we carried out a paired t-test with numbers before and after as a treatment. There were significantly fewer visits per year post-construction (overall before mean = 2.76, overall after mean = 1.69, t=2.37, df=36.25, p=0.02). We used the mean number recorded per visit (see below) but the imbalance between before and after wind farm construction data collection is more likely to result in fewer records after construction (due to fewer visits). This contrasted with the pattern shown in the data (see results).

Surveyors used visual scanning and audible sounds to detect black grouse, carefully counting birds while making an effort not to double count if birds were flushed. The location and behaviour of each black grouse was marked on a map and the grid reference recorded. Across all sites there were 64 survey years; in 29 survey years there was one visit and in 17 survey years there were two visits to the site.

All observations were digitised in GIS and their distance to the nearest turbine was calculated in GIS.

1.3.4 Analysis

A - Number of black grouse

We compared the abundance of lekking males before wind farm construction with the abundance of lekking males after construction. Each year males gather at lek areas and display to attract a mate. Counting birds at these lek areas has been an established survey method and is a way to monitor population changes in this species. Although we report the results from the analysis performed only on

A) Lek area	B) No of years data collected before construction	C) No of years data collected post-construction	D) Survey area	E) Notes
1a	0	5	Known lek was visited	
1b	1	8	Known lek was visited	
1c	0	4	Known lek was visited	Lek moved due to change in farming management and was subsequently no longer monitored.
2	2	2	500 meters around the proposed wind farm	
3	2	7	500 meters around the proposed wind farm and the habitat management area south of the wind farm	
4	2	3	200-300 meters around the proposed wind farm and the planned track south of the wind farm	
5	1	3	Known leks were visited	

Table 4: Overview of the data collection at the seven study sites. The number of years of data collection before or after construction (columns B and C) were not necessarily in consecutive years. We identified nine lek areas for analysis: groups of birds found more than 1.5 km apart were regarded as separate lek areas. Site 1 and 7 were therefore split up into three lek areas each. (Continued on next page)

A) Lek area	B) No of years data collected before construction	C) No of years data collected post-construction	D) Survey area	E) Notes
6	3	4	500 meter around the proposed wind farm track. This survey area included 60 turbines of the total 140 turbines on the site	
7a	7	2	Known lek was visited	Data was also collected for 1 year between the construction of the original wind farm and the extension
7b	6	3	Known lek was visited	Data was also collected for 2 years between the construction of the original wind farm and the extension
7c	5	3	Known lek was visited	Data was also collected for 2 years between the construction of the original wind farm and the extension

Table 4: Overview of the data collection at the seven study sites. The number of years of data collection before or after construction (columns B and C) were not necessarily in consecutive years. We identified nine lek areas for analysis: groups of birds found more than 1.5 km apart were regarded as separate lek areas. Site 1 and 7 were therefore split up into three lek areas each.

lekking males, analyses based on all sightings (regardless of the bird's behaviour or sex) were very similar and did not change our results (data not shown). For the analysis we used the total number of lekking males per visit for each year in the breeding season for each site. Data were skewed and would not transform to a Gaussian distribution and so we used a Poisson error distribution to obtain a Gaussian distribution in the residuals. We used Bayesian generalized linear mixed models (GLMMs) with a Poisson error distribution. We used lek area as a random effect as there were large differences in number of birds lekking between the lek areas (for example lek area 2 had around two birds lekking while lek area 7a had between 10 and 30 birds lekking). We used a Bayesian approach because it accounted for the overdispersion that we found when fitting a Poisson model using conventional frequentist methods. To our knowledge there is no accurate way to account for high overdispersion in these models using conventional frequentist methods.

We identified nine lek areas for analysis: groups of birds found more than 1.5 km apart were regarded as separate lek areas. Sites 1 and 7 were therefore split up into three lek areas each. Lek area 1b and all the lek areas of site 7 were included in the analysis as separate lek areas. However, there were insufficient data from lek area 1a and 1c before construction and therefore these lek areas were excluded from the analysis.

We ran the model with three chains for 65000 iterations. We found that autocorrelation within the chains was reduced sufficiently when every 50th iteration was saved (thinning rate of 50). In addition, the chains converged after 15000 iterations; therefore these first 15000 iterations of the Markov chains were discarded (burn-in). This resulted in a sample size of 1000 saved iterations from which our posterior estimates were drawn.

B - Distribution of black grouse

To determine whether the distribution of lekking males changed in relation to the construction of the wind farm we performed Bayesian GLMMs with Gaussian error with lek area as a random effect. We used a total of nine lek areas

in the analysis (the same lek areas as in section above). From each black grouse individual or, when black grouse were within 100m from each other, from each group of black grouse we calculated the distance in meters to the nearest wind turbine. These values were then square-root-transformed so the residuals from the model conformed to a Gaussian distribution. We ran the models with three chains for 13000 iterations with a thinning rate of 10 and a burn-in of 3000, resulting in a sample size of 1000.

In addition, to investigate whether the distribution of lekking males only changes relatively near to the planned wind farm we split data into two groups: (1) lek areas where the median distance of the black grouse to the nearest planned wind turbine before construction was less than 937m (937m was the median distance when it was calculated from all recorded distances of black grouse to the nearest wind turbine before construction across all lek areas); (2) lek areas where the median distance of the black grouse to the nearest planned wind turbine before construction was above this value. The median distance of the black grouse to the nearest planned wind turbine before construction in group 1 was 250m and in group 2 1380m. Before construction, over 80% of the black grouse leks in group 1 were within 500m of the nearest wind turbine, while almost 80% of the black grouse leks in group 2 were over 900m away. We then performed the above Bayesian GLMM on these data with site as a random effect. Group 1 included data collected at lek areas 2, 4, 5 and 6 and group 2 included data from the other five lek areas (1c, 3, 7a, 7b and 7c). For the model we ran three chains for 65000 iterations with a thinning rate of 50 and a burn-in of 15000, resulting in a sample size of 1000.

All statistical tests were performed in R (version 3.0.0) (R Core Team, 2013). The GLMMs were performed with package MCMCglmm version 2.17 (Hadfield, 2010). For all Bayesian GLMMs convergence was assessed by a graphical check of adequate mixing of the three chains as well as using the Gelman-Rubin diagnostic. We did not specify any priors and therefore the models used the default priors. This means that the priors for the fixed effects were centered on 0 and had large variance and for the variance components a flat improper prior was used. Results were regarded as significant when the 95% credibility interval

of the posterior distribution excluded 0.

1.4 Results

Black grouse were recorded at seven different sites both before and after the construction of wind farms of varying sizes. Males continued to lek at six of the seven sites (Figure 2)

1.4.1 A - Changes in abundance of black grouse following wind farm construction

Overall, the abundance of lekking males did not change significantly after the wind farms were constructed (mean = -0.03, 95% CI = -0.51 – 0.48) (see Table 5 and Figure 2). There was strong support for variability between lek areas as the posterior distribution of the random effect term 'lek area' was centered well away from zero (mean = 1.20, 95% CI = 0.47 – 4.56). In addition, the random effect term 'lek area' explained the majority of the variation in the data (mean = 63%, 95% CI = 39-87%). This is not surprising given the difference in the number of lekking males between the lek areas.

1.4.2 B - Distribution of black grouse

The distribution of lekking males was further away from the wind farm after construction than before, across all sites (mean = 3.16, 95% CI = 0.57 – 5.86). As the data were square-root-transformed this means that the average square root distance between the lekking males and turbines increased by 3.16. It is difficult to interpret the exact quantity of this figure because back-transformation of the data is not possible. However, as an approximate guide the median distance of lekking males from wind turbines before construction was 937m and after was 1331m.

When we split the data into two groups based on those that were initially closer to the turbines before construction, only the distribution of lekking males in group 1 changed (group 1 mean = 7.51, 95% CI = 2.78 – 12.24, group 2 mean = 0.94, 95% CI = -2.13 – 4.29, see also Table 5 and for the raw data Figure 3). Again as

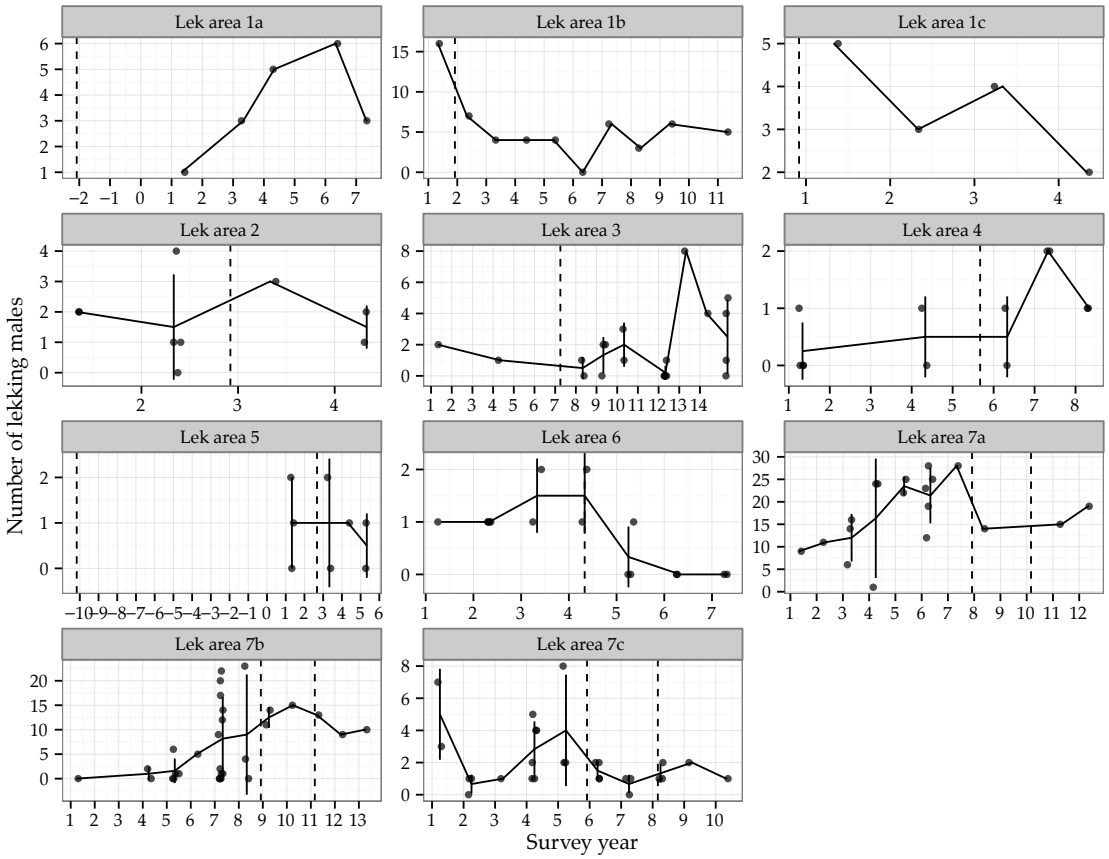

Figure 2: The number of lekking male black grouse per site per year. The line is the mean number of lekking males in each year, the dots show the total number of lekking males per visit. The dashed line indicates the year of the construction of the wind farm or extension (in case of a second dashed line). The survey years are different calendar years for each lek area. For example, survey year 2 for lek area 1a does not correspond to the same calendar year as survey year 2 for the other lek areas. The sites are all in southern Scotland and the last national survey in 2005 found that black grouse especially in southern Scotland were still in decline (Sim et al., 2008). The habitat modifications on the sites consisted of tree planting (sites 1, 3, 4, 5 and 7), predator control (culling of foxes and crows) (site 1 and 7), grazing restrictions (site 1 and 7), removal of redundant fences (site 7), marking of existing fences (site 7), tree felling (site 3, 6 and 7), creating small wet areas (site 3 and 7) and blanket bog restoration (site 3 and 6) in addition to the construction of the turbines.

Model	Term	Mean	95% lower	95% upper
Abundance of lekking males	Intercept	0.60	-0.32	1.36
	Before/after	-0.03	-0.51	0.48
	Random effect	0.63	0.39	0.87
Distribution all lekking males	Intercept	30.67	21.83	40.93
	Before/after	3.16	0.57	5.86
	Random effect	0.79	0.63	0.96
Distribution lekking males group 1	Intercept	17.87	11.63	24.20
	Before/after	7.51	2.78	12.24
	Random effect	0.23	<0.01	0.74
Distribution lekking males group 2	Intercept	39.60	28.34	49.01
	Before/after	0.94	-2.13	4.29
	Random effect	0.71	0.42	0.96

Table 5: 95% credibility intervals of the posterior distribution from the different models run via MCMCglmm. The table shows values of the intercept and the fixed effect. In addition, it also gives the proportion of data that is explained by the random effect. For example: if looking at the first model (abundance lekking males) it shows that there was no change in the abundance after the wind farm was constructed as the 95% credibility interval overlapped with zero; the random effect mean shows the proportion of the deviance or variance (depending on model) explained by the lek area effect, in this case on average 63% of the variation was explained by differences between lek areas.

an approximate guide the median distance of lekking males from wind turbines in group 1 before construction was 250m and after was 803m. For group 2 the median distance of lekking males from wind turbines before construction was 1380m and after was 1624m.

1.5 Discussion

This study demonstrates that data from EIA studies and post-construction monitoring can be used to answer important scientific questions and provide an evidence base for land management decisions. Using data from seven sites collected by a range of companies we have been able to show that black grouse

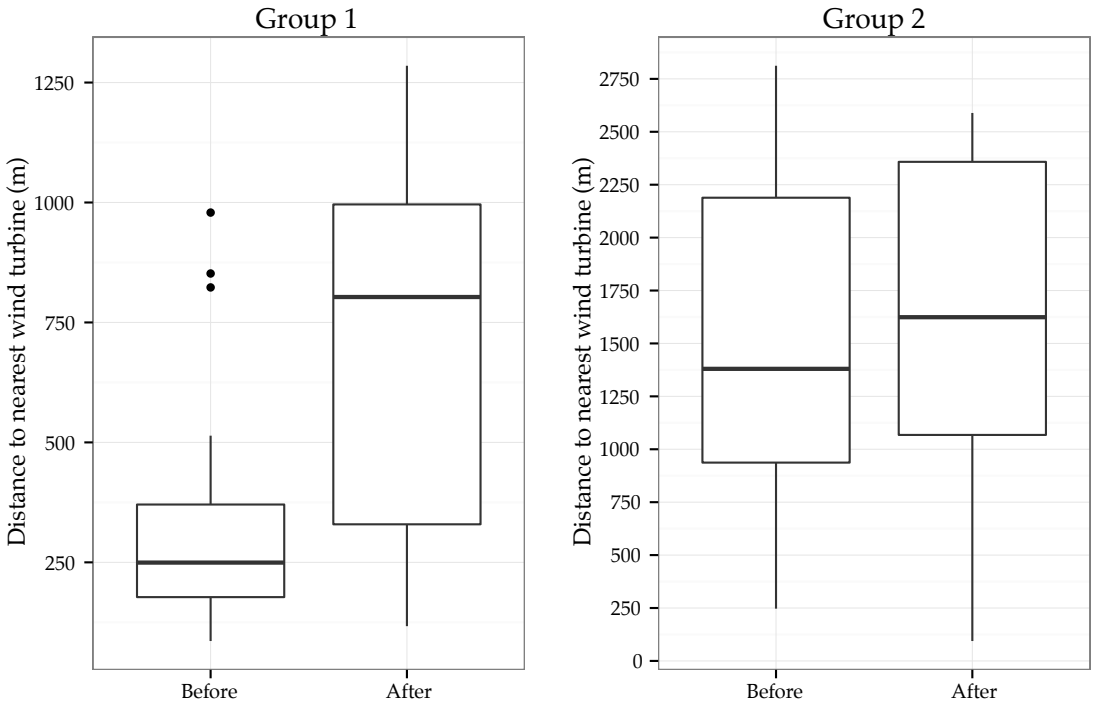

Figure 3: A boxplot of the distance from lekking males to the nearest wind turbine. Group 1 includes the lek areas where the mean distance of black grouse records was below 937m (n = 4 lek areas, left plot), Group 2 includes the sites where the mean distance of black grouse records was above 937m (n = 5 lek areas, right plot). On the x-axis of each plot the data before wind farm development are displayed on the left and on the right the data after wind farm development.

persisted on all seven wind farm sites for the duration of our study period, which ranged between two and fifteen years post-construction. While, across all sites, the abundance of black grouse did not change in the areas around the wind farms, we did find evidence that black grouse leks generally within 500m of the nearest planned wind turbine moved locally after construction.

Currently, conservationists assess the potential effects of wind farms on black grouse largely on expert opinion and potentially taking into account the one peer-reviewed scientific study in the Austrian Alps that showed that black grouse left the wind farm site (Zeiler and Grünschachner-Berger, 2009). In contrast to that study, our results indicate that the abundance of black grouse was not significantly affected by wind farms. This might be contrary to expectations as a decrease in the abundance of black grouse at wind farm sites in Scotland might even be expected

regardless of the wind farm development. Firstly, due to the fact that the British black grouse populations have declined during the period of the study. The last national survey in 2005 found that black grouse in Britain, especially in southern Scotland, were still in decline (Sim et al., 2008). Secondly, because small leks seem more susceptible to change (Geary et al., 2012) and therefore a change in black grouse numbers would first be expected at the lek areas with observed low numbers. However, we found that even at these lek areas (lek area 2, 3, 4, 5, 6 and 7c) we did not observe consistent declines. This coupled with the fact that the majority of the variation in the data was explained by the difference between the lek areas (the random effect) and black grouse are still in decline in the region, strengthens the case that the abundance of black grouse is not affected by wind farms over the duration of our study. However, it could be that we did not find a significant change in black grouse abundance due to mitigation measures in the form of habitat enhancement aimed to benefit black grouse at six out of the seven sites. Although it is interesting to note that black grouse persisted on all seven sites so even where no mitigation occurred black grouse populations remained extant.

We found some evidence of localised movement by black grouse leks which were generally within 500m of the proposed turbines. The cause of movement of lek sites could be driven by several causative factors. These include the operation of wind farm infrastructure, volume of visitors, changes in land management both within and surrounding the site, and habitat enhancement measures designed to attract black grouse into specific areas. As black grouse require a range of habitat throughout the year which includes heathland, young and open forests, pastures and meadows (Bernard, 1981; Picozzi and Hepburn, 1986; Baines, 1994; Angelstam et al., 2000), any change in the distribution of black grouse could be due to one or more of these factors mentioned above.

Even though we found that black grouse leks moved at some sites, it does not necessarily mean that they did not use the wind farm area at all. At one of the study sites additional records were obtained during the winter months and any evidence of black grouse presence such as feathers, droppings and footprints were recorded during all surveys at this site. These extra records indicated frequent

use of the area within 500 meters of the turbines and occasional use of the area underneath the turbines (data not shown). Further study is needed to determine how much and when black grouse use the wind farm area.

Our results could have implications for planning applications as the information to assess potential effects of a planned wind farm on black grouse is limited. For example, the SNH guidance stipulates that all leks within 1.5km of a proposed onshore wind farm should be surveyed (SNH, 2010) and the RSPB buffered black grouse leks by a similar distance when producing its sensitivity guidance and maps in 2008 (Bright et al., 2008). Warren and Baines (2004) suggested suitable habitat should extend for 200-500 ha from a lek (which corresponds to 0.8 - 1.3 km buffer) and Cayford (1993) suggested that 500 - 700 ha may be enough to sustain one lek (which corresponds to 1.3 km and 1.5 km buffer). Our results show that black grouse found generally within 500m of planned turbines moved locally after construction. Therefore, detectable effects may be defined within 500m of planned turbines. However, we would suggest survey boundaries up to 1.5km from wind turbines to gather information on local population change, especially if EIA and post-construction data from multiple sites are to be used for this species.

1.5.1 Insights into use of EIA data

Our study provides an important example of how data that are gathered during EIAs and post-construction monitoring can be used to inform wider issues of potential effects of wind farms on wildlife and in turn inform planning decisions. In addition, we show how researchers and developers could work together to provide an evidence base for land management decisions. Nonetheless, there are limitations with this type of data. Firstly, the data are not readily accessible. We had to contact six different wind farm developers as there is no central repository for this type of data. However, only one contact replied and provided data sets. Recently, there has been effort in Scotland to create a central repository for data from EIAs and post-construction monitoring by the Scottish Windfarm Bird Steering Group (www.swbsg.org, 2013). This is positive but ideally there should be a data repository across many countries so scientists and consultants could access this data for research and planning decisions. This

would in turn mean that scientists can carry out studies that would include all sites where their species of interest is occurring and therefore make more robust conclusions about potential effects.

A second limitation of this data is that the proximate cause of an effect at a single site is not possible because the data are observational. However, if these data were more widely available this would be less of an issue.

Thirdly, survey effort differed between sites and between years which introduces variance. More consistency would have provided more power to detect effects of interest. For example, in our case the number of visits reduced after construction, and therefore if we had found a decline in the abundance of black grouse at wind farms we would not have known if this would have been due to reduced survey effort or if there really is a decline. A greater consistency in survey effort would make the data more useful, without necessarily increasing the cost or difficulty of data collection.

Chapter 2. A model to predict the effect of wind turbine noise on avian communication

2.1 Abstract

Anthropogenic noise has been shown to impact avian distributions and their breeding success as it affects their ability to communicate. Most literature has focused on urban and traffic noise but here we explore the impacts of wind turbine noise. The number of wind farms are rising rapidly and are often located away from urban areas and thus may affect a different range of species. However, the impacts of wind farm noise on birds are largely unknown. Here we present a mathematical model which calculates the decrease in area where an avian acoustic signal can be heard and interpreted by a receiver after development of a wind farm. The model predicted significant reductions in the communication ability of species that call at frequencies below 2 kHz. In addition, our model shows that other species vocalising above 2 kHz can also be affected but not as severely as species vocalising below 2 kHz. We use our model with data on a range of bird species. We found severe reductions of active space were predicted for black grouse, wheatear and goldcrest and to a lesser extent for lapwing, golden plover, skylark, blackbird and willow warbler. Some of these species have previously been shown to avoid wind farm areas, indicating that noise may play an important role in causing bird disturbance by wind farms. In addition, our results show that the masking effects of turbine noise depend on the turbine type and wind speed. Our method can be applied to any bird species and to novel wind turbines with known noise spectra and we provide the code to do so. During an Environmental Impact Assessment, this model could easily be adapted for the species and site in question, and the predictions from the model could be used to guide turbine placement to minimize noise disruption for birds.

2.2 Introduction

Many animals use acoustic communication. Birds in particular use acoustic signals for example to attract mates, deter rivals, or to warn conspecifics of predators. According to the acoustic adaptation hypothesis, most of these signals would have evolved in such a way that the transmission of the signal is optimum for the habitat that the bird is in (Morton, 1975). The ability to communicate is limited by the area where a signal can be perceived by a receiver over a background noise of a particular habitat. The area where such a signal could be both heard and interpreted is called active space (Marten and Marler, 1977; Brenowitz, 1982; Klump, 1996; Lohr et al., 2003). Many factors influence the transmission of a signal and therefore also its active space; such as humidity, temperature, ambient pressure, background noise levels, topography, amplitude and frequency of the signal, among others (Marten and Marler, 1977; Wiley and Richards, 1978; Forrest, 1994; Blumenrath and Dabelsteen, 2004). For a receiver to be able to detect a signal, the signal must be louder than the background noise of the receiver's habitat. However, for a receiver to be able to discriminate and identify the song or call from other signals, a higher signal-to-noise ratio is required than for simple detection. When there are increased noise levels in a particular environment, for example due to anthropogenic noise sources, the hearing threshold of the receiver is increased and in turn the active space of a signal is reduced. This means that the signal needs to be louder to be detected in a noisy environment than in a quiet environment. This masking threshold can be more than 25 dB higher than the threshold in quiet conditions (Klump, 1996; Lohr et al., 2003).

Many studies have focused on these masking effects of urban or traffic noise (Slabbekoorn and Peet, 2003; Rheindt, 2003; Brumm, 2004b; Fuller et al., 2007; Nemeth and Brumm, 2009; Verzijden et al., 2010; Parris and McCarthy, 2013). Slabbekoorn and Peet (2003) found that great tits (*Parus major*) in urban environments have increased the minimum frequency of their song compared with great tits in rural environments. Other studies have found similar effects for blackbirds (*Turdus merula*), chiffchaffs (*Phylloscopus collybita*), robins (*Erithacus*

rubecula) and others (Nemeth and Brumm, 2009; Verzijden et al., 2010; McLaughlin and Kunc, 2013). Other strategies to overcome the masking effect of noise include increasing the amplitude of the song or changing the timing of singing. For example, nightingales (*Luscinia megarhynchos*) have been found to increase their song's amplitude (Brumm, 2004a), and robins in urban environments switched to singing at night, depending on the noise levels during the day (Fuller et al., 2007). It has been suggested that increasing the minimum frequency is not very beneficial as it only marginally increases the active space of the signal (Nemeth and Brumm, 2010; Parris and McCarthy, 2013). However, increasing the amplitude of the signal, greatly increases the active space of the signal (Nemeth and Brumm, 2010; Parris and McCarthy, 2013). Recently, Nemeth et al. (2013) have suggested that an increase in minimum frequency is merely a by-product of increasing the amplitude of the signal due to vocal constraints; higher frequencies are easier to produce at higher amplitudes.

Furthermore, various studies have found reduced bird densities and reduced breeding success at roadsides possibly due to noise (Reijnen et al., 1996; Rheindt, 2003; Parris and Schneider, 2009; Halfwerk et al., 2011). However, other confounding variables associated with roads, such as edge effects and increased fatalities through collision with vehicles, might also explain observed reductions in bird densities around roads. In order to distinguish between these possibilities, McClure et al. (2013) introduced traffic noise into a roadless environment by placing speakers along a line; the resulting reduction in bird densities was therefore attributable to noise.

To predict which species might be affected by urban noise, Parris and McCarthy (2013) created a mathematical model which calculates the decrease in active space of bird acoustic signals after moving from quiet forest habitats to noisy urban habitats. They found that the active space of signals below 3 kHz would be reduced by more than 27%, with the greatest reduction of 63% for those species that vocalise around 1.5 kHz. The active space of other signals would be reduced by around 20%.

However, noise sources other than from an urban environment also have the

potential to affect the natural environment. In this paper, we focus on wind turbine noise for two reasons. First, the impacts of noise from wind turbines on birds are not well understood. Several studies have found reduced bird densities at wind farms (Leddy et al., 1999; Larsen and Madsen, 2000; Hötker et al., 2006; Fijn et al., 2007; Madsen and Boertmann, 2008; Pearce-Higgins et al., 2009; Zeiler and Grünschachner-Berger, 2009; Bevanger et al., 2010; Garvin et al., 2011; Dahl et al., 2012; Pearce-Higgins et al., 2012). However, the underlying causal mechanisms for this reduction remain unclear. It is speculated that wind farm noise might affect birds' communication and that, therefore, they avoid the areas near the turbines but there is no direct evidence to support this (Leddy et al., 1999; Drewitt and Langston, 2006; Hötker et al., 2006; Madders and Whitfield, 2006; Zeiler and Grünschachner-Berger, 2009). However, there is some evidence from mammals; the vigilance behaviour of the California ground squirrels (*Spermophilus beecheyi*) was altered most probably due to turbine noise (Rabin et al., 2006).

Second, the number of wind farms worldwide has been increasing rapidly in the last 15 years (GWEC, 2013) and so it is important to predict their impacts on birds, where 13% of all species are threatened with global extinction (Hoffmann et al., 2010). The use of wind farms will continue to increase and some projections predict that by 2020 20% of the electricity worldwide will be provided by wind energy (IPCC, 2012). Wind farms are usually built away from urban areas, and thus often in quiet environments. Therefore, the effect of wind turbine noise might be relevant to a range of species that are not subjected to urban or traffic noise. As such, predictions of noise propagation from wind farm sites could be very valuable as a planning tool to evaluate and minimize the impacts of wind farms on birds.

In this paper, we investigate and predict which signal frequencies are affected by wind turbine noise and whether this effect depends on turbine type and wind speed. We illustrate the method with vocalization data from thirteen bird species but the method can be used for any species or turbine and we have made the code openly available. Several of the bird species we have selected are known to be affected by wind farms. The model we present here is adapted from Parris

and McCarthy (2013). Their model calculates the reduction of active space of acoustic signals after moving from a quiet environment to noisy urban habitats and the noise in the urban habitat is homogeneous. Wind turbine noise comes from a point source (the turbine itself) and its intensity decreases with increasing distance from the turbine. We therefore adapted the model presented by Parris and McCarthy (2013) to account for noise coming from a point source.

2.3 Methods

2.3.1 Overall approach

Wind turbine noise comes from a particular point source which is the turbine itself and propagates through the environment with its intensity decreasing with increasing distance from the source. This means that the masking effect that turbine noise has, depends on the location of the sender and the receiver. We modelled the potential masking effect of turbine noise for several locations of the sender in relation to the turbine (10m, 50m, 100m, 150m, 200m, 250m, 300m, 350m, 400m, 500m, 600m, 800m, 1000m, 1200m and 1400m from the turbine) (see below for details of noise profiles for different turbines and different wind speeds). For each of these locations we calculated the active space (that is the area where a signal can be heard and interpreted) in an environment with turbine noise and without turbine noise by creating a grid of receiver locations which were separated by 10m, and calculating the intensities of a signal and the turbine noise for each of these locations. We then calculated whether a signal could still be detected and discriminated at each of these receiver locations in an environment with and without turbine noise. Using this information, the reduction of active space of a signal by turbine noise was calculated (for an example of the model outputs see Figure 4). We calculated the reduction of active space for frequencies from 0.25 to 8 kHz as well as for thirteen specific species (see Table 6).

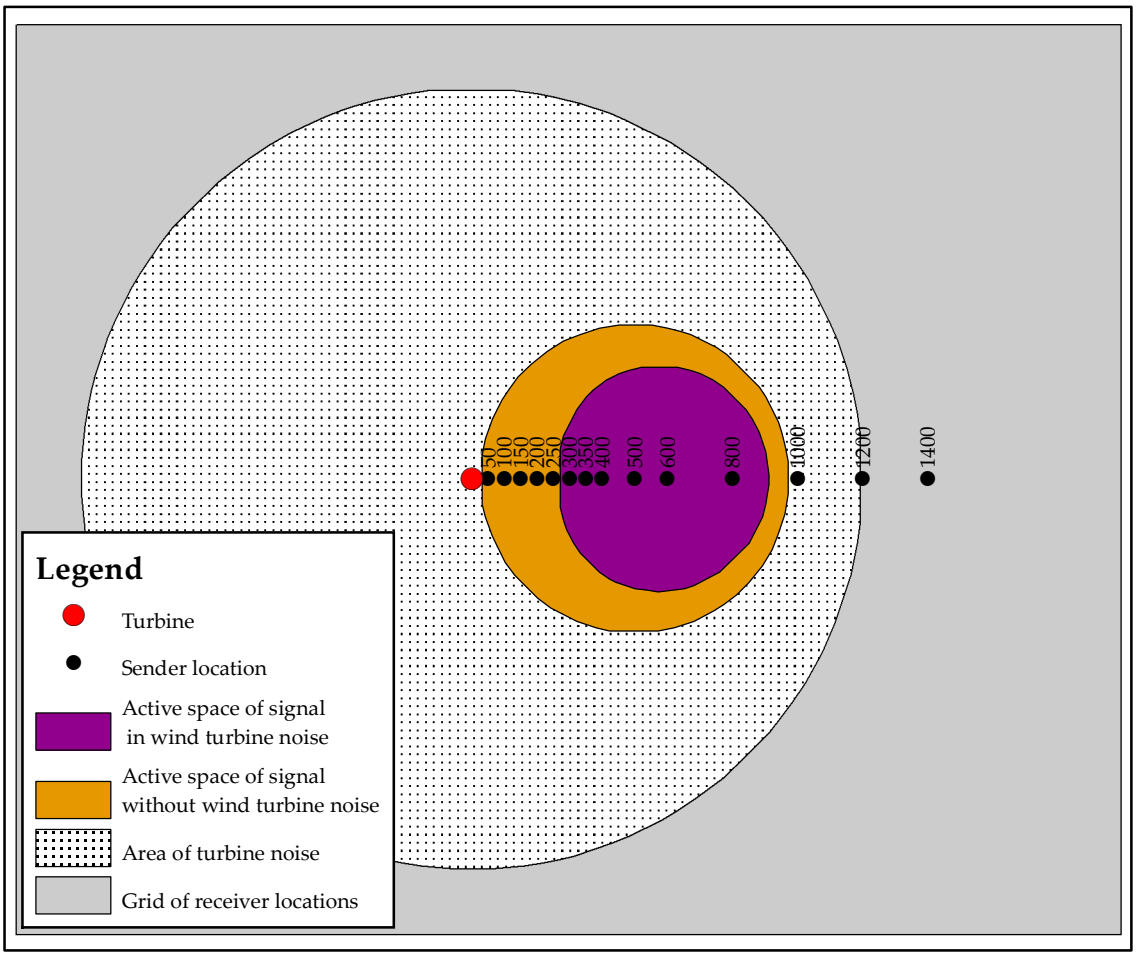

Figure 4: Example of model output for a 0.5 kHz signal broadcast at 500m from the wind turbine. The location of the wind turbine is shown by a red dot and the grid of potential receiver locations is shown in grey. The model calculated where the wind turbine noise could be heard (dotted area). The model also predicted the active space of a 0.5 kHz signal broadcast at 500m from the wind turbine in absence (orange area) and presence (purple area) of wind turbine noise. Additional sender locations used in our model are shown by black dots.

2.3.2 Model description

Sound pressure level of a signal at receiver's location

As a sound wave propagates spherically, the energy of a sound spreads over an ever-increasing surface. The inverse square law predicts that the amplitude of a sound originating from a source will decrease by 6 dB for each doubling of distance from the source (Forrest, 1994). However, this is only true under ideal conditions with no atmospheric absorption, scattering and/or boundary interferences. Therefore, in natural environments a sound generally attenuates at

a faster rate due to excess attenuation (such as atmospheric absorption, etc). The difference in amplitude of a sound between the sound source and a receiver location (DD) can be calculated by using the inverse square law and correcting for excess attenuation (adapted from Lohr et al., 2003; Nemeth and Brumm, 2010; Parris and McCarthy, 2013):

$$DD = 20\log_{10}\left(\frac{d_1}{d_2}\right) + \frac{E(d_1 - d_2)}{100} \quad (1)$$

Where d_1 is the distance from the receiver to the sound source, d_2 is the distance from the sound source at which the amplitude was measured (1m in our case) and E is the excess attenuation in dB/100m. As excess attenuation is in dB/100m, we divided the second term by 100 as d_1 and d_2 are per meter. We then deducted DD from the amplitude at the sound source to get the amplitude of a signal at the receiver. This is different from the model presented by Parris and McCarthy (2013), where they solved for d_1 (maximum communication distance in their case) and in their model DD was the difference between amplitude of the signal at d_2 and the masked threshold of the receiver. Due to propagation of wind turbine noise, the amplitude of wind turbine noise decreases with increasing distance from the wind turbine. Therefore, the masked threshold of the receiver depends on the receiver's location in relation to the wind turbine. In order to calculate the masked threshold of the receiver, we calculated the amplitude of wind turbine noise at each receiver location in the grid by calculating the difference in amplitude between the wind turbine and the receiver location (DD). We also calculated the amplitude of a signal between 0.25 and 8 kHz at each receiver location in the grid.

Signal-to-noise ratio

For the receiver to be able to discriminate a signal, it must be louder than the masking threshold of the receiver. This threshold is the background noise plus the critical ratio or signal-to-noise ratio (Langemann et al., 1995; Klump, 1996). Therefore, background noise of a particular habitat will influence the masked threshold of the receiver. The louder the background noise, the higher the masked threshold is and thus the distance over which a signal can be communicated can

be significantly reduced. In addition, the signal-to-noise ratio might vary for different frequencies and generally is larger for higher frequencies (Lohr et al., 2003). To be able to determine whether the receiver would be able to discriminate a signal, we added the average signal-to-noise ratios as analysed by Klump (1996) in ten different bird species to the spectrum level of wind turbine noise to get the masked threshold of different frequencies in wind turbine noise (see Figure 5). For signals in an environment without turbines, we determined whether a signal was louder than the masked threshold of background noise. For signals in an environment with turbines, we first checked which masked threshold was higher (the turbine or background masked threshold) then we determined whether a signal was louder than this highest masked threshold.

Reduction of active space

For each receiver location in the grid we calculated whether a signal could be heard and interpreted in an environment with and without turbine noise (see below for how the grid was constructed). To calculate the total area where a signal could be heard and interpreted, we converted the grid of receiver locations to polygons by using the 'Minimum Bounding Geometry' tool in ArcGIS and selecting the option 'Convex hull' and grouping the receiver locations based on common values in the attribute table. The attribute table of the grid contained information for each receiver location whether the signal could be heard and interpreted in an environment with and without turbine noise. We then calculated the area of each polygon. The reduction in active space was calculated as follows:

$$100 - \left(\frac{A_{turbine}}{A_{background}} * 100\% \right) \tag{2}$$

Where $A_{turbine}$ is the active space of a signal with turbine noise and $A_{background}$ is the active space of a signal without turbine noise. The active space was calculated for each location of the sender (10m to 1400m from the turbine) and for the different signal frequencies (see model inputs below).

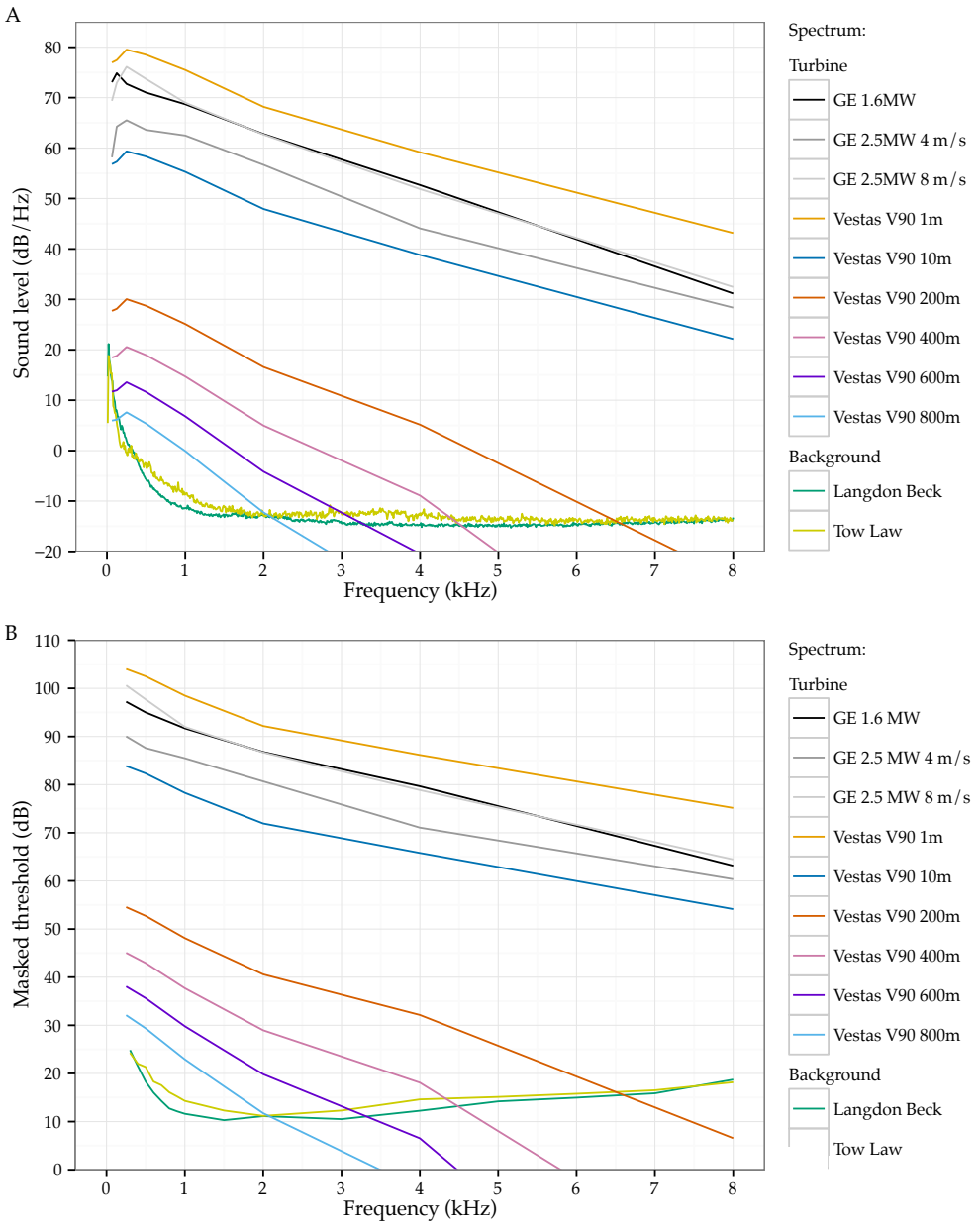

Figure 5: Spectrum level of noise (A) and masked threshold of signals (B) for three different wind turbine models and two different habitats. The three different turbine models were: GE 1.6MW, GExl 2.5MW and Vestas V90 3MW. Spectrum levels were from 1m of the sound source except for Vestas V90 for which 10m, 200m, 400m, 600m and 800m from the sound source are also included. The spectrum levels from the turbines were obtained from the manufacturers. The background noise profiles of Langdon Beck and Tow Law were calculated in Avisoft SasLab Pro and the levels were then averaged for every 10 Hz. For the receiver to be able to discriminate a signal, it must be louder than the masking threshold of the receiver. The masked thresholds are the spectrum noise levels and the critical ratios taken from Klump (1996) added together. With increasing levels of noise, the higher the masked threshold is and thus the distance over which a signal can be communicated can be significantly reduced.

Model assumptions

To keep the model relatively simple, we assumed a flat, homogeneous surface. In addition, we assumed that from each sender location there was one individual calling and that there was one turbine in the landscape.

2.3.3 Model inputs

Grid of receiver locations, sender and turbine location

The grid of receiver locations was created in ArcGIS by using the 'Create Fishnet' tool and selecting the option 'Create Label Points' with a separation of 10m between each location. The size of the grid was large enough so that the active spaces of the different signals were completely inside the grid (3.4 by 2.8km). In addition, a turbine location was created inside the grid with sender locations at aforementioned distances from the turbine.

Signal inputs 0.25 to 8 kHz

We modelled the active space of frequencies between 0.25 and 8 kHz. As frequency of bird vocalizations is correlated with body mass (Wallschläger, 1980; Gillooly and Ophir, 2010) we were able to estimate the body mass for each of these frequencies by using the following equation (from Parris and McCarthy, 2013):

$$F = am^{-b} \tag{3}$$

The parameters a and b were taken from Parris and McCarthy (2013) (their Table 1) and F is the frequency in kHz and m is mass in grams.

As mass also correlates with the power of a signal, the following equation was used to estimate signal power P (mW) (from Parris and McCarthy, 2013):

$$P = Am^{B} \tag{4}$$

The parameters A and B were taken from Parris and McCarthy (2013) (their Table 1) and m is mass in grams.

Signal power can be converted to sound pressure level (SPL) using the following equation, derived from the reference value of 0.03 W = 97 dB SPL (Brackenbury, 1978) (from Parris and McCarthy, 2013):

$$L = 82 + 10\log_{10}(P) \tag{5}$$

Here signal power P is in mW and the sound pressure level L is the amplitude of a signal in dB at 1m from the sender.

Signal inputs specific species

We selected 13 different species so that a wide frequency range was included. Furthermore, this selection was based on species that were either reported as being displaced by wind turbines (black grouse (*Tetrao tetrix*), golden plover (*Pluvialis apricaria*) and wheatear (*Oenanthe oenanthe*) (Zeiler and Grünschachner-Berger, 2009; Pearce-Higgins et al., 2009, 2012)) or by noise (great tit (*Parus major*), blackbird (*Turdus merula*) and robin (*Erithacus rubecula*) (Slabbekoorn and Peet, 2003; Brumm, 2004a; Montague et al., 2012; McLaughlin and Kunc, 2013)). For each species we used the peak frequency (that is the frequency with the highest amplitude) and the amplitude of the song in our model. We used these signal features as it is currently unknown how other signal features such as bandwidth, signal length, syllable rate, amplitude modulation, and the use of complex elements such as trills affect signal degration and propagation (Lohr et al., 2003; Parris and McCarthy, 2013). For certain species the amplitude and peak frequency was known from literature, for six other species the amplitude was calculated via above equation 4 and 5. For golden plover and oystercatcher (*Haematopus ostralegus*) we were not able to obtain the peak frequency from the literature and therefore we used recordings archived at Xeno-canto (http://www.xeno-canto.org/; catalogue numbers: XC25075, XC37498, XC42344, XC53817, XC57855, XC57857, XC76241, XC76502, XC85842, XC90437). We then measured the peak frequency of these species via spectral analysis using the software package Praat 5.2.34 and averaged the measurements for all songs for each species. The peak frequencies for black grouse and robin were measured in the same way in Praat from own recordings that were made at

Species	Scientific name	Peak frequency (kHz)	Amplitude (dB SPL at 1m)	S/N ratio (dB)[j]	Mass (g)	Habitat and song height	Predicted EA (dB/100m)[l]
Black grouse	*Tetrao tetrix*	0.454[a]	103.5[e]	24.00	1200[k]	OF ground	23.19
Curlew	*Numenius arquata*	1.680[b]	101.1[e]	23.75	742[b]	OF 10m	2.66
Golden Plover	*Pluvialis apricaria*	2.530[c]	94.9[e]	24.50	209[k]	OF 10m	3.12
Lapwing	*Vanellus vanellus*	1.850[b]	95.0[e]	23.75	211[b]	OF 10m	2.71
Oystercatcher	*Haematopus ostralegus*	3.025[c]	99.6[e]	25.00	546[k]	OF 10m	3.43
Skylark	*Alauda arvensis*	3.954[d]	90.8[f]	27.00	n/a	OF 10m	3.99
Wheatear	*Oenanthe oenanthe*	3.790[b]	84.3[e]	26.75	24[b]	OF 1m	14.72
Blackbird	*Turdus merula*	2.340[b]	87.4[g]	24.25	n/a	DEC 2m	7.82
						DEC 10m	12.37
Goldcrest	*Regulus regulus*	7.300[b]	75.0[h]	30.00	n/a	CON 2m	21.31
						CON 10m	13.17
Great tit	*Parus major*	4.030[b]	88.8[g]	27.00	n/a	DEC 2m	12.20
						DEC 10m	21.29

Table 6: Model inputs for specific species. (Continued on next page)

Species	Scientific name	Peak frequency (kHz)	Amplitude (dB SPL at 1m)	S/N ratio (dB)[j]	Mass (g)	Habitat and song height	Predicted EA (dB/100m)[l]
Robin	*Erithacus rubecula*	3.620[a]	90.0[h]	26.00	n/a	DEC 2m	11.03
						DEC 10m	18.91
Song thrush	*Turdus philomelos*	3.530[b]	99.7[h,i]	26.00	n/a	DEC 2m	10.90
						DEC 10m	18.65
Willow warbler	*Phylloscopus trochilus*	4.000[b]	76.5[h,i]	27.00	n/a	DEC 2m	12.12
						DEC 10m	21.13

Table 6: Model inputs for specific species. Signal-to-noise ratios (S/N ratio) are taken from Klump (1996). We took the species' habitat and song height and predicted the excess attenuation (EA) via piece-wise linear regression using data from Marten and Marler (1977). The peak frequency and amplitude for each species were either taken from literature or calculated via their mass. The different habitats are: OF = open field, DEC = deciduous forest, CON = coniferous forest. For example, black grouse sings at a peak frequency of 0.454 kHz at a predicted amplitude of 103.5 dB on the ground in an open field habitat. The predicted EA for black grouse is 23.19 dB/100m. [a]Calculated via own recordings. [b]Data from Hu and Cardoso (2009). [c]Calculated via recordings taken from Xeno-canto (http://www.xeno-canto.org/). [d]Data from Linossier et al. (2013). [e]Calculated via mass. [f]Data from Briefer et al. (2008). [g]Data from Nemeth and Brumm (2010). [h]Data from Brackenbury (1979). [i]Data from Parris and McCarthy (2013). [j]Data from Klump (1996). [k]Data from Robinson (2005). [l]Based on data from Marten and Marler (1977).

Langdon Beck in UK (coordinates: 54°41'N, 2°13'W) on 26th -28th of May 2012 and Chopwell Woods in UK (54°55'N, 1°47'W) on 20th-23rd of April 2013 respectively. See Table 6 for full details on mass, frequency and amplitude for each species.

Excess attenuation

Our model takes excess attenuation (see above) into account by using data collected by Marten and Marler (1977). For modelling the active space for frequencies between 0.25 and 8 kHz we used data collected by Marten and Marler (1977) for open field and deciduous forest with leaves at 2m height for the signals and 10m height for turbine noise. For specific species we used data on excess attenuation for their respective habitat (e.g. for songbirds we used deciduous forest with leaves at 2m height and 10m height) (see Table 6 for full details). These data were then used in a piece-wise linear regression to model excess attenuation for each of the different habitats used. Each model was used to predict excess attenuation for the different frequencies (see Table 6 for predicted excess attenuation and supplementary material for the piece-wise linear regressions).

Noise spectra

For all signals modelled in an environment without any turbine noise we used the noise spectra obtained from recordings at a site consisting of open field habitat: Langdon Beck in Teesdale (coordinates: 54°41'N, 2°13'W) in May 2012. For signals modelled with turbine noise we used the noise spectra obtained from manufacturers. To determine whether the masking effect of wind turbine noise is dependent on the turbine type, we modelled the effects of turbine noise for two different wind turbine types in 10 m/s wind: Vestas V90 3MW and GE 1.6MW. Currently, Vestas and GE hold the top positions in the global wind turbine manufacturer market share rankings (MAKE Consulting, 2014). Furthermore, we investigated whether the masking effect of wind turbine noise is dependent on wind speed by modelling the masking effect of turbine type GE 2.5xl in 4 m/s

and 8 m/s wind. In addition to these, we also checked reduction of active space of turbine type Vestas V90 3MW using a different background noise spectra that was created from recordings at a different open field habitat site, but these did not change the results of our model (see Figure S2 for model outputs). These recordings were made at Tow Law in County Durham (coordinates: 54°45'N, 1°49'W) in June 2012. For the model run for thirteen specific species we used the background noise spectra of Langdon Beck and turbine noise spectra from type Vestas V90 3MW. We used twelve different recordings for the background noise spectra at Tow Law and ten for the background noise spectra at Langdon Beck. These recordings were made with SM2+ bioacoustic recorder (Wildlife Acoustics Inc.) with an omni-directional microphone. The recordings were taken on different days and throughout the dawn period to get a good representation of noise levels when birds are most active. The noise spectra of the recordings were analysed in Avisoft SasLab pro. After noise spectra were created, the spectra were calibrated to their respective amplitude levels that were measured via SongScope 4.1.3A.

2.4 Results

2.4.1 Change in active space for 0.25-8 kHz

For turbine type Vestas V90 3MW in an open field, the active spaces for frequencies below 2 kHz are predicted to be most affected by wind turbine noise, with up to 88% (95% confidence interval: 31-100%) reduction when the sender is near the turbine and still up to 53% (95% confidence interval (CI): 22-69%) when the sender is at 500m from the turbine. Higher frequencies are predicted to be reduced by up to 100% (95% CI: 0-100%) if the sender of the signal is within 10m of the turbine, the active space is predicted to be less affected at 250m with up to 39% (95% CI: 15-54%) reduction for signals at 4 kHz and no reduction (95% CI: 0-0%) for signals at 8 kHz. At 500m from the turbine, the reduction drops further to 20% (95% CI: 14-21%) for signal at 4 kHz. At around 2 kHz our model predicted that there is no effect of wind turbine noise on active space in an open field habitat. However, the 95% CI suggests that a reduction up to 19% could be expected if the sender is between 250m and 800m from the turbine (see Figure 6).

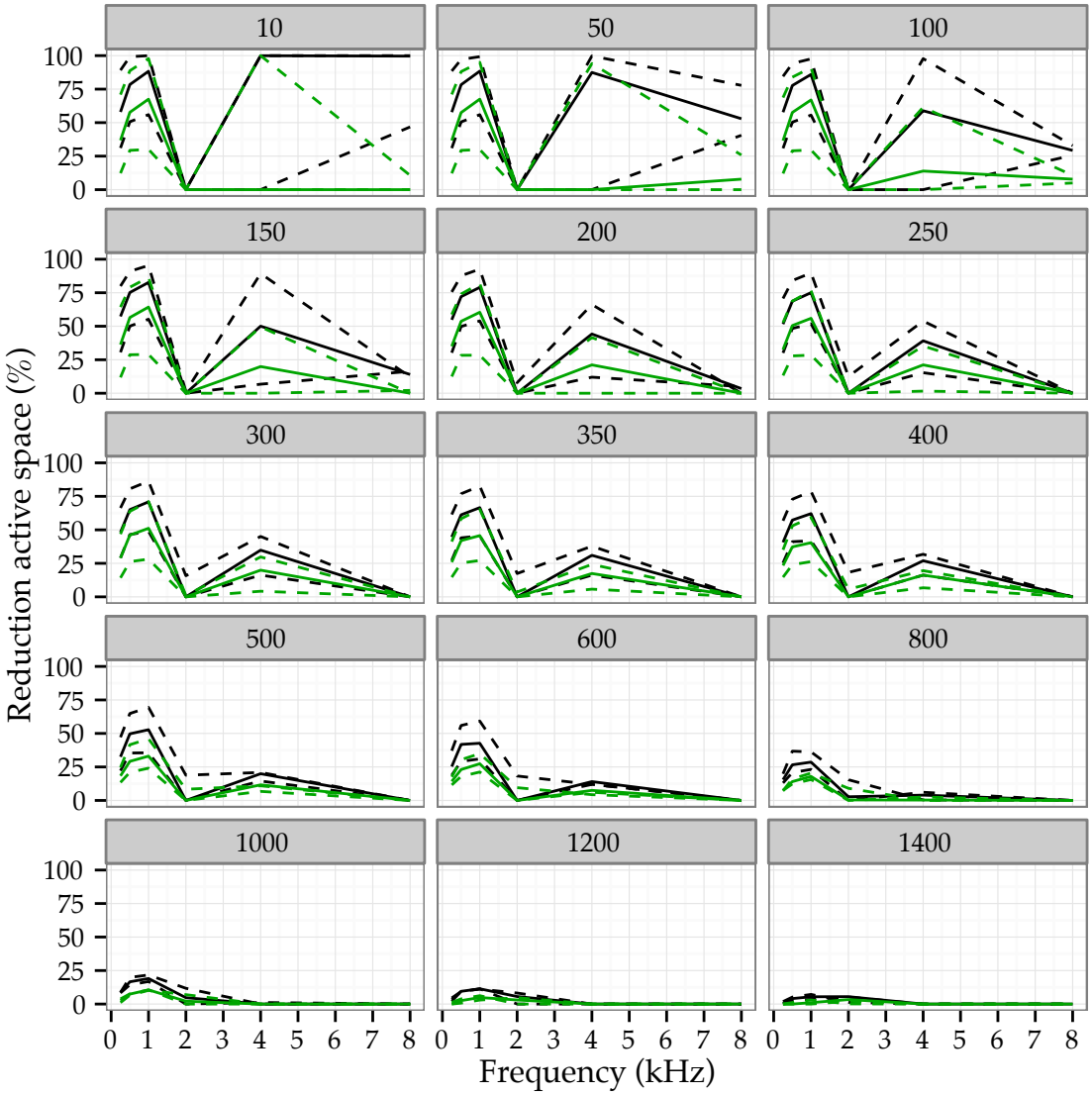

Figure 6: Reduction of active space after construction of different turbine models in an open habitat. In black: Vestas V90 3MW and in green: GE 1.6MW. The dashed lines denote the 95% confidence intervals.

2.4.2 The masking effect of different turbine types

For the 1.6MW turbine type from GE, the model predicted less reduction of active space overall than for the turbine type V90 3MW from Vestas (see Figure 6). However, the pattern looked very similar: substantial reduction for the lower frequencies and no reduction for signals at 2 kHz and medium reduction for signals above 2 kHz. The model predicted on average no reduction of active space for frequencies above 2 kHz when the sender is at 10m from the GE 1.6MW wind turbine, although there was some uncertainty in this for signals at 4

kHz, as the 95% CI did include maximum reduction of 100%. If the sender is between 100m and 500m from GE 1.6MW turbine, a reduction between 0-21% (95% CI: 0-61%) is predicted for frequencies above 2 kHz. When the sender is more than 500m away from the GE 1.6MW turbine, a reduction of 0-8% in active space is predicted for frequencies above 2 kHz.

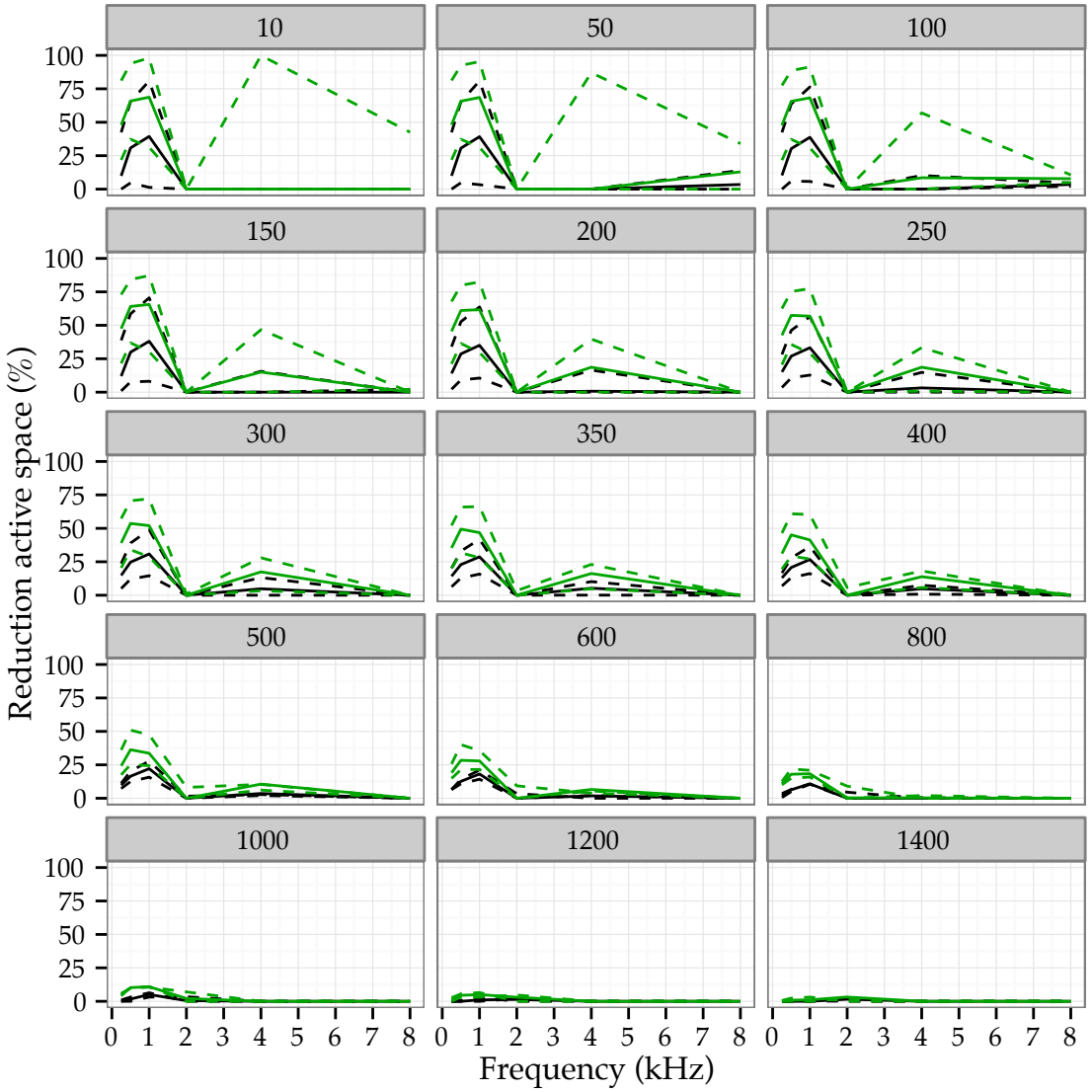

Figure 7: Reduction of active space after construction of turbine model GExl 2.5MW in different wind speeds in an open habitat. In black: 4 m/s and in green: 8 m/s. The dashed lines denote the 95% confidence intervals.

2.4.3 The masking effect of different wind speeds

The predicted reduction of active space at 8 m/s wind speed is higher overall than the predicted reduction at 4 m/s wind speed (see Figure 7). The mean predicted

reduction is between 9-39% between 10 and 500m for frequencies below 2 kHz for 4 m/s, and between 24-69% for 8 m/s. For frequencies between 2 kHz and 8 kHz, the mean predicted reduction of active space is between 0-5% for 4 m/s and between 0-19% for 8 m/s.

2.4.4 Change in active space for individual species

Open field species

Our model predicted that the communication for two open field species will be affected quite severely; these were black grouse and wheatear with reductions between 81-100% (95% CI: 75-100%) and 57-100% (95% CI: 49-100%) respectively if the sender is up to 250m from the wind turbine and reductions between 60-77% (95% CI: 54-84%) and 11-43% (95% CI: 11-52%) respectively if the sender is between 300-500m from the wind turbine (see Figure 8). For golden plover, lapwing and skylark our model predicted a lower but still quite substantial reduction of active space (golden plover 10-250m: 0-17% (95% CI: 0-100%) and 300-500m: 19-20% (95% CI: 0-41%); lapwing 10-250m: 0-29% (95% CI: 0-100%) and 300-500m: 27-29% (95% CI: 3-52%); skylark 10-250m: 0-19% and 300-500m: 14-19%). For oystercatcher and curlew, the model predicted less reduction in active space compared to the other wader species, although the reduction for curlew did have large confidence intervals especially if the sender is near the turbine (oystercatcher 10-250m: 0% (95% CI: 0-24%) and 300-500m: 3-9% (95% CI: 2-24%); curlew 10-250m: 0-3% (95% CI: 0-99%) and 300-500m: 7-14% (95% CI: 0-41%)).

Woodland species

When the sender is vocalising at 2m height, the active spaces of willow warbler and goldcrest are predicted to be substantially reduced with 28% reduction for willow warbler if the sender is at 50m from the wind turbine and 97% reduction for goldcrest if the sender is 10m from the wind turbine (see Figure 9 solid dots). When the sender is further away from the turbine the active space is predicted to be reduced less (e.g. at 200m willow warbler: 8% and goldcrest: 2%). The active

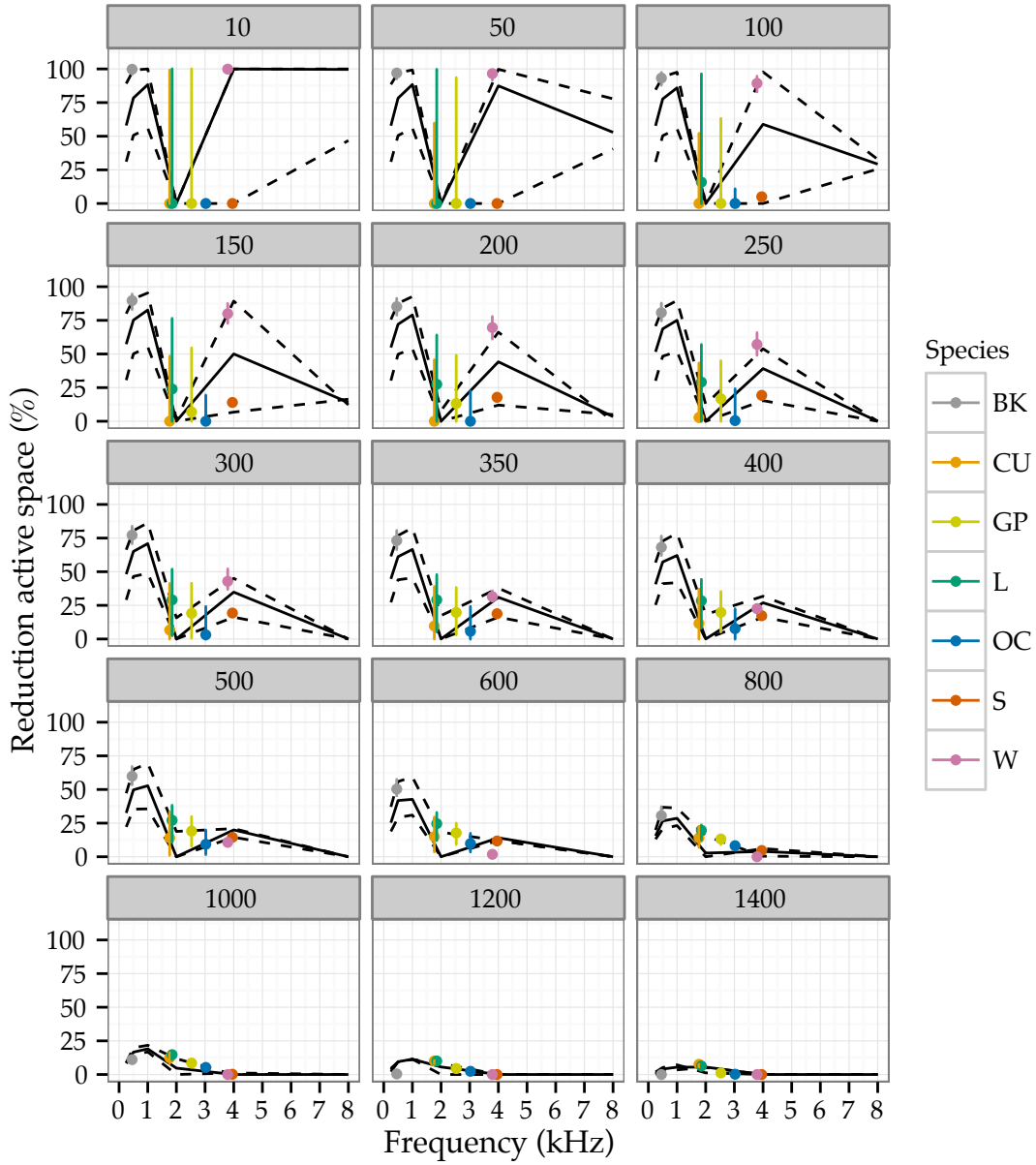

Figure 8: Reduction of active space after construction of a turbine in an open habitat for different species. The lines indicate the prediction of reduction of active space after construction of turbine model Vestas V90 3MW in an open habitat. The dashed lines denote the 95% confidence intervals. The error bars around the points denote the 95% confidence intervals. Species included in the model were: "BK" = Black grouse*, "CU" = Curlew*, "GP" = Golden plover*, "L" = Lapwing*, "OC" = Oystercatcher*, "S" = Skylark and "W" = Wheatear*. An asterisk (*) denotes for which species the amplitude is estimated from mass. The predictions in active space for these species have a 95% confidence interval, as the amplitude estimation had lower and upper bounds. For details about the model inputs for these species see Table 6.

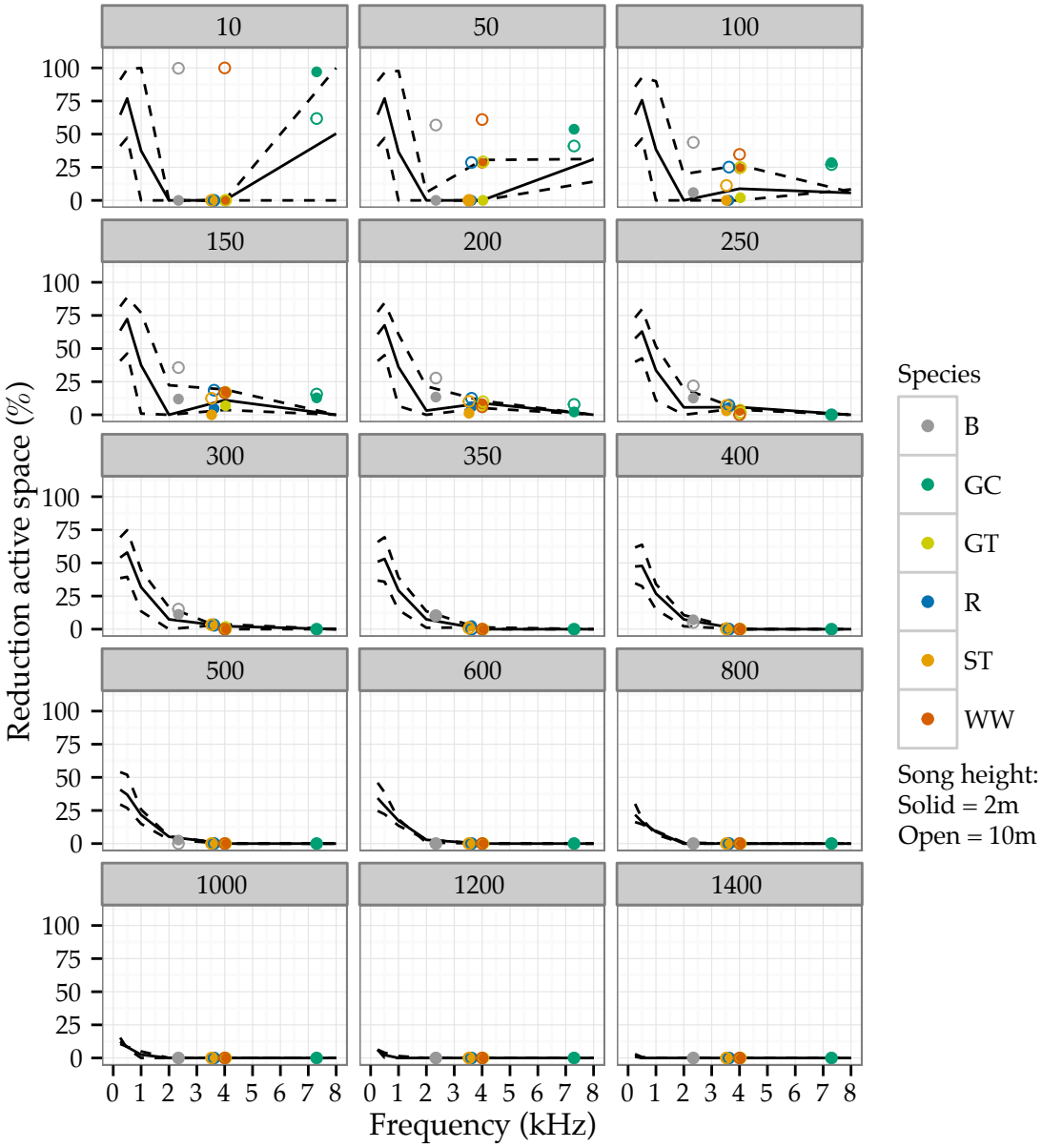

Figure 9: Reduction of active space after construction of a turbine in a deciduous forest for different species. The lines indicate the prediction of reduction of active space after construction of turbine model Vestas V90 3MW in a deciduous forest. The dashed lines denote the 95% confidence intervals. The solid circles show results from models of excess attenuation at 2m height, open circles at 10m height. Species included in the model were: "B" = Blackbird, "GC" = Goldcrest, "GT" = Great tit, "R" = Robin, "ST" = Song thrush and "WW" = Willow warbler. For goldcrest, models were run with excess attenuation in coniferous forest.

spaces of the other woodland species are less affected according to the predictions of our model (maximum reduction: blackbird 13% at 200m, song thrush 3% at 250-300m, robin 6% at 200m and great tit 7% at 150-200m).

Interestingly, the model predicted a greater reduction of active space if the sender is in deciduous woodland at 10m height compared with at 2m height. Especially for willow warbler (shown with red circles in Figure 9) and blackbird (shown with grey circles in Figure 9) this either means a complete reduction of its active space or no reduction at all if the sender is near the turbine (see open and solid circles in Figure 9). In addition, the active spaces of robin and great tit were also predicted to be substantially more reduced at 10m height than at 2m height. In contrast, the model showed the reverse in coniferous forest: the reduction of active space of goldcrest (shown with darkgreen circles in Figure 9) is predicted to be less at 10m height than at 2m height (see open and solid circles in Figure 9).

2.5 Discussion

Our model predicted which avian vocal signals will be affected by wind turbine noise. The model predicted that species communicating at frequencies below 2 kHz will have significant reductions in their abilities to communicate with conspecifics (reductions in active space) but this is dependent on the turbine type and wind speed. For black grouse, a species we examined in this range, the model predicted quite severe reductions. In addition, our general model showed that other species vocalising above 2 kHz can also be affected but not as severely as species vocalising below 2 kHz. Severe reductions of active space were also predicted for wheatear and goldcrest and to a lesser extent for lapwing, golden plover, skylark, blackbird and willow warbler.

Most of the predictions of reduction of active space for individual species were within the 95% confidence intervals of the general model, showing that our general model can be used to predict the reduction of active space that is due to wind turbine noise. However, when predicting the effects for an individual species which are vocalising around 2 kHz, it will be more valuable to input the amplitude or mass of the species as this will give a better indication of whether

these species may be affected. For example, the peak frequencies for curlew and lapwing are around 2 kHz and the general model predicted no to low reductions around this frequency, indicating that these species will not be affected. However, the prediction from the model for the specific species had wide confidence intervals, suggesting that these species could be disturbed by wind turbine noise but this depends on the amplitude they vocalise at.

The wide confidence intervals in the models, in the general model and in the models for particular species where their amplitudes were unknown, were due to the large differences between the lower and upper bounds of the estimated amplitude for a particular frequency or species. For example, the amplitude estimations for curlew and lapwing were 92.9-109.1 dB and 87.7-102.1 dB respectively. Obtaining information on the amplitude that the species of interest vocalises at, will improve the prediction of whether a particular species could be affected by wind turbine noise. This amplitude information needs to be collected in the field as the information is not available from the literature.

2.5.1 Reduction of active space dependent on turbine type and wind speed

We found that the reduction of active space is dependent on turbine type and wind speed. This is hardly surprising as the sound pressure level of turbine noise changes with turbine type and wind speed. For example, for the turbine type GE 2.5xl examined here the sound pressure level ranges from 95.7 in 4 m/s wind speed to 104.2 dB in 8 m/s wind speed. As wind speed increases the sound pressure level of turbine noise will increase and our predictions show that this impedes acoustic communication further. However, wind speed will also increase ambient noise levels as wind-induced vegetation noise levels are proportional to the wind speed, therefore wind turbine noise could be potentially masked by ambient noise levels (Bolin, 2009; Bolin et al., 2010). Further research is needed to investigate how this might affect bird communication. Furthermore, the turbine type Vestas V90 3MW is 6.3 dB louder than GE 1.6MW, and the predictions for these turbine types show that the Vestas V90 3MW will cause more reduction of active space the the GE 1.6MW.

2.5.2 *Potential explanation for lower densities at wind farms*

Our model predicted severe reductions for black grouse, wheatear and goldcrest and modest reductions for lapwing, golden plover, skylark, blackbird and willow warbler. It is quite probable that these species will avoid wind farm areas due to high noise levels. Especially if these species cannot adjust their vocalisation to overcome the masking effect of wind turbine noise, for example by increasing the peak frequency or amplitude of their song. Several studies have shown that two of these species, black grouse and wheatear, avoid areas near wind farms (Zeiler and Grünschachner-Berger, 2009; Pearce-Higgins et al., 2009, 2012) and some studies have suggested that this might be due to high noise levels at wind farm sites as it might affect their communication (Leddy et al., 1999; Drewitt and Langston, 2006; Hötker et al., 2006; Madders and Whitfield, 2006; Zeiler and Grünschachner-Berger, 2009). The model predicted that their communication will be affected, which means it is likely that wind farm noise is an important factor in causing their displacement. Further study is needed to investigate if other factors also play a role.

Currently, it is unknown whether golden plover, lapwing and skylark are avoiding wind farm areas, as there have been mixed results from various studies (reviewed by Hötker et al., 2006). In addition, Pearce-Higgins et al. (2009) report no avoidance of wind farm areas for lapwing and skylark, although the effect for skylark was approaching significance ($p=0.057$). In the same study, golden plover was found to avoid the wind farm, whereas other studies observed no avoidance behaviour by golden plover (Douglas et al., 2011; Fielding and Haworth, 2013). Further investigation is needed to determine whether the masking effect of wind farm noise on the communication of these species is causing population-level effects, for example through reduced breeding success. We also show that our model predicted minor reductions in active space for curlew, a species that avoids wind farm areas (Hötker et al., 2006; Pearce-Higgins et al., 2009). However, the predictions from our model had large confidence intervals, especially near the wind turbine. It would, therefore, be very valuable to obtain amplitude information for this species as this would improve the prediction in

reduction of active space for this species and thereby aid the assessment of whether wind turbine noise is a contributing factor causing their avoidance.

Previous studies have shown that disturbance effects for passerines are smaller in magnitude than for other species, operating usually over 100-200m (Leddy et al., 1999; Hötker et al., 2006; Devereux et al., 2008; Pearce-Higgins et al., 2009). The predictions from our models for woodland species are consistent with this as most of the masking effects are predicted within this distance.

Our model predicted that goldcrest, robin, blackbird, willow warbler and great tit will be affected by wind turbine noise, but this depends on the height at which the species sings. However, no research to date has investigated the effect of wind farms on these species. Great tits, blackbirds and robins are known to be affected by urban noise; they have been shown to increase their minimum frequencies in urban areas or in response to urban noise (Slabbekoorn and Peet, 2003; Nemeth and Brumm, 2009; Montague et al., 2012; McLaughlin and Kunc, 2013) and great tits also have reduced breeding success near roads (Halfwerk et al., 2011). Therefore, species that are predicted to be affected by wind turbine noise in this study, either the individual species examined or species that vocalise at low or high frequencies, might avoid wind farm areas and/or have reduced breeding success near wind farm areas.

2.5.3 Potential disturbance effects of wind farms underestimated

Currently many studies investigate whether birds are disturbed by wind farms by looking into the distribution of birds at wind farm areas. We identified here that wind farm noise could play an important role in the avoidance of wind farm by birds. However, noise has been shown to affect breeding success as well (Habib et al., 2007; Francis et al., 2009; Halfwerk et al., 2011) and this may go unnoticed as breeding densities may remain high due to inexperienced or low quality birds occupying noisy areas (Reijnen and Foppen, 1991; Habib et al., 2007), lower predator rates in noisy areas (Francis et al., 2009) or due to fewer competitors for food (Slabbekoorn and Halfwerk, 2009). Therefore, the disturbance effects of wind farms on birds may be underestimated as breeding

numbers give little insight into the mechanisms by which birds are affected (Halfwerk et al., 2011). It is thus important to focus on the effects of wind farms on different elements of reproductive success to be able to understand whether and how birds are affected by wind farms.

2.5.4 Song post at 2m or at 10m height

As the forest species we included in our model could be singing between 2m and 10m height or over, we ran the model for both of these situations. Our model predicted that the active space of a signal broadcast at 10m height in a deciduous forest is reduced more than a signal broadcast at 2m height. In contrast, a signal broadcast in a coniferous forest would have less reduction at 10m height than at 2m height. This is due to their respective excess attenuation – more excess attenuation at 10m height in a deciduous forest and more excess attenuation at 2m height in a coniferous forest (Marten and Marler, 1977) (see also Table 6). Due to the increase in excess attenuation, the signal will lose energy more quickly and therefore the turbine noise will have more of a masking effect. It is therefore more beneficial for birds to sing at lower heights in a deciduous forest and at higher heights in a coniferous forest and by doing this they might avoid the masking effect altogether. For example, our model predicted no reduction of active space for willow warbler and blackbird if they sing at 2m height, but if they sing at 10m height their ability to communicate is severely impeded.

2.5.5 Model assumptions

For our model, we assumed a flat homogeneous surface, which often is not the reality. A wind farm might be built in an environment with different types of habitat and changes in altitude. We also assumed that there would be one turbine in the landscape. However, often wind farms consist of multiple turbines and this would increase the overall sound level generated by the wind farm and therefore bird communication might be affected more than predicted here. In addition, as with the model presented by Parris and McCarthy (2013), which our model is based on, a number of signal features were not included in the model but might affect signal propagation or signal discrimination. Signals that contain

features such as a wide frequency bandwidth or amplitude modulation could require a higher signal-to-noise ratio for detection and discrimination than pure tones (Lohr et al., 2003). In addition, it is unknown how other signal features such as signal length, syllable rate and the use of complex elements such as trills affect signal propagation and degradation. Further research is needed so that these signal features could be included in the model. Furthermore, we did not account for changes in humidity, temperature, ambient pressure and other weather conditions, which can change during the day and between seasons and also affect signal propagation. In future, the model could be adapted to include these signal features and account for a heterogeneous habitat.

2.5.6 Model is easily applicable to any situation

Our model can account for different wind turbine types by inputting a different noise spectrum, which is easily obtainable from the manufacturer. In addition, our model can easily be adapted for different species of interest if their amplitude and peak frequency is known. If their song amplitude is not known, this can then be calculated via the mass of the species. Peak frequency can be measured via recordings, either gathered online via song banks or gathered in the field. Furthermore, our model can account for different background noise levels at a specific site than our site as the background noise profile used in our model can easily be changed. To do this, recordings need to be taken at the site in question and from these the noise profile can be created. In this way, predictions for different weather conditions can also be examined.

2.5.7 Application to wind farm planning

As our model can give a good indication as to which species might be affected by wind turbine noise, it could in turn help planning decisions and aid conservation efforts. During an Environmental Impact Assessment, this model could easily be adapted for the species and site in question, and the predictions from the model will give an indication whether the species is to be affected by the development. This model will be especially valuable for species where the disturbance effects of wind farms are unknown or in question.

In addition, this model could be used to identify the best turbine type as explorations can be made regarding which turbine type will cause the least communication disturbance for birds. Furthermore, our model could give insight to which location would give the least disturbance by noise. For example, it could aid the placement of the turbines by investigating the distance that wind farm noise would have negligible effects. As well as turbine location it could also help placing the wind farm itself as different models could be run for each location based on its respective background noise spectrum. However, when the background spectra are very similar, the predictions will differ by only a small amount (see Figure S2). But if locations are investigated where background noise levels differ substantially, the noisier location might be favoured as it might mask the wind farm noise. For example, locations near roads could be very favourable as road noise has already been identified to mask wind turbine noise for humans (Pedersen et al., 2010). Further studies are needed to investigate the masking effects of road noise on turbine noise for birds.

2.5.8 Conclusion

Our model gives a good indication whether avian communication will be affected by the development of a wind farm. It can be applied to any bird species and can be adapted for any wind turbine type. We have identified that noise plays an important role in causing bird disturbance by wind farms. Furthermore, other sources of anthropogenic noise have been shown to cause reduced breeding success and densities. Therefore, by predicting the effects of wind turbine noise, our model could aid wind farm planning decisions and help conservation effects.

2.6 Supplementary material

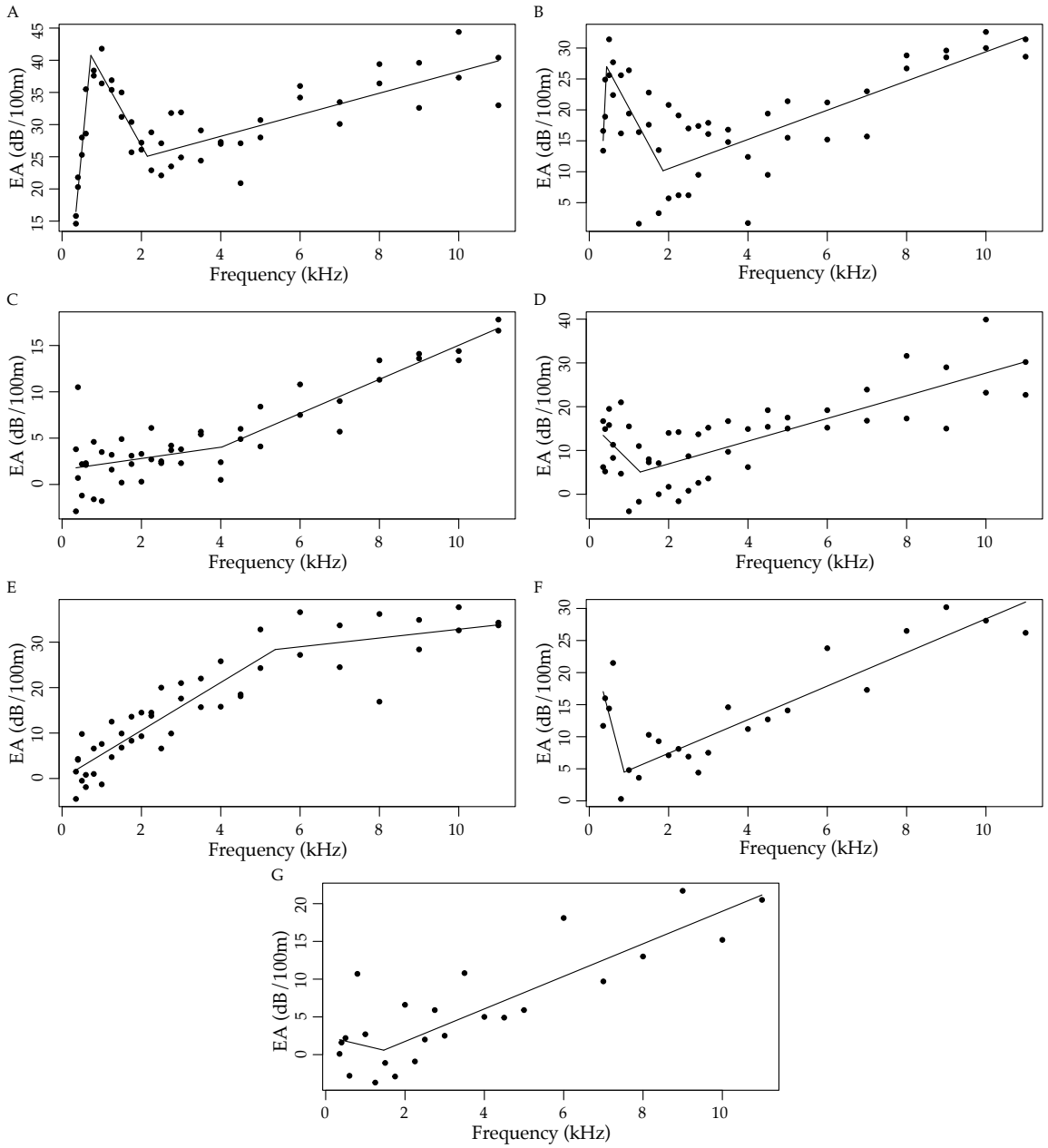

Figure S1: Excess attenuation (EA) of a signal (dB/100m) in various habitats broadcast at different heights: (A) open field ground; (B) open field 1m; (C) open field 10m; (D) deciduous forest 2m; (E) deciduous forest 10m; (F) coniferous forest 2m; (G) coniferous forest 10m. In natural environments a signal generally loses energy at a faster rate than the inverse square law predicts due to excess attenuation caused by atmospheric absorption, scattering and/or boundary interferences. Piece-wise linear regressions (lines) using data from Marten and Marler (1977) (points).

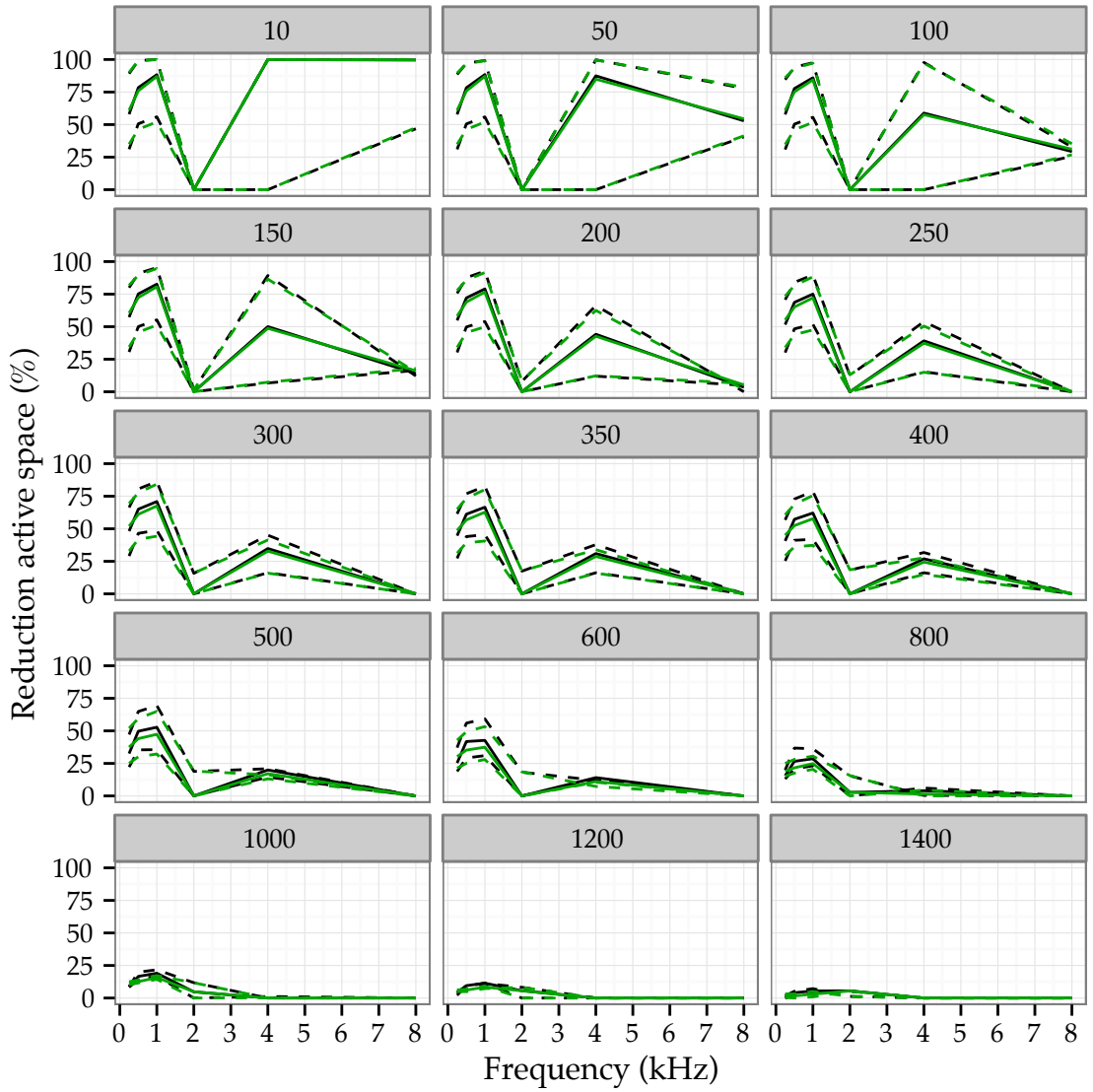

Figure S2: Reduction of active space after construction of turbine model Vestas V90 3MW in two different open habitats. In black: Langdon Beck (coordinates: 54º41'N, 2º13'W) and in green: Tow Law (coordinates: 54º45'N, 1º49'W). The dashed lines and error bars around the points denote the 95% confidence intervals.

Chapter 3. Wind farm noise affects territorial defence behaviour in European robins (*Erithacus rubecula*)

3.1 Abstract

Many species rely on acoustic signals to communicate messages which are critical to survival. However, noise from anthropogenic sources, such as traffic, industry or wind farms, could impede the transmission of these signals. While many studies have looked at the effects of anthropogenic noise on avian communication, the effects on territory defence are currently unknown. We investigated whether wind turbine noise, a source of anthropogenic noise often generated in remote locations away from urban areas and main roads, affects territory defence in the European robin (*Erithacus rubecula*). We found that European robins used low frequency song elements to signal aggression and thereby avoid the need for costly physical conflict. In contrast, in the presence of wind farm noise European robins substantially reduced their usage of low frequency song elements during simulated intrusion, indicating that they responded with a much reduced aggressive signal. Robin males may have sacrificed signal efficacy for signal detection due to the masking effects of wind turbine noise. This may affect their ability to deter an intruder, which may lead to expenditure of extra time and energy and could lead to increase risks of injury and as a consequence reduced breeding success. To our knowledge, this is the first study that shows that territory defence is affected by anthropogenic noise. The effect that anthropogenic noise has on territory defence may be one of the important causes of the reduced breeding success found near noisy areas, including wind farm areas.

3.2 Introduction

Many species rely on acoustic signals to communicate messages which are critical to survival. However, noise from anthropogenic sources, such as from traffic, industry or wind farms, could impede the transmission of these signals. Birds in particular depend on acoustic signals and the masking effect of anthropogenic noise could have direct fitness consequences (Slabbekoorn and Ripmeester, 2008). For example, the ability of males to defend a territory which has food provision, shelter and nesting opportunities is dependent on acoustic signals (Ten Cate et al., 2002; Brumm, 2004a; Ripmeester et al., 2007; Brumm and Ritschard, 2011). In addition, acoustic signals could encode species' identity and individual quality, which females use to find and choose a mate (Riebel, 2003). Anthropogenic noise could also mask alarm, contact and begging calls, which are all crucial for survival (Mathevon, 1997; Leonard and Horn, 2005; Leavesley and Magrath, 2005). If we are to predict with accuracy the effect of anthropogenic noise on birds then a thorough understanding of the impacts on behaviour and life history are key. In this study we present evidence of disruption by wind farm noise on key behaviours likely to be linked to life history traits.

To overcome the masking effect of anthropogenic noise, birds have been found to increase the amplitude of their song (Brumm, 2004b) or change the timing of their singing to periods when it is quieter (Fuller et al., 2007). Furthermore, they may change the frequency of their song depending on the acoustic signature of noise (Halfwerk and Slabbekoorn, 2009). Most of the energy of traffic or urban noise is contained within the lower frequencies and many species have been found to increase the minimum frequency of their song in response to such noise (Slabbekoorn and Peet, 2003; Verzijden et al., 2010; Ripmeester et al., 2010; Montague et al., 2012; McLaughlin and Kunc, 2013). Anthropogenic noise has also been linked to reduced breeding densities and reproductive output (Habib et al., 2007; Bayne et al., 2008; Francis et al., 2009; Halfwerk et al., 2011), especially for species which are vocalising at lower frequencies (Goodwin and Shriver, 2011).

Low frequency songs can play an important role in territory defence as they can

signal aggression and are used in threatening displays (Morton, 1977; Benedict et al., 2012). Males can win contests by signalling superior fighting ability and motivation to attack an intruder, thereby avoiding the need for costly physical conflict (Maynard Smith and Price, 1973). In the presence of anthropogenic noise, birds may sacrifice this signal efficacy for signal detection. As a consequence, a resident male might not be able to successfully deter competitors, which may result in the expenditure of extra time and energy and could also lead to increase risks of injury.

Currently, the effects of anthropogenic noise on territory defence are unknown. In this study, we aim to identify if and how territory defence is affected by wind turbine noise, a source of anthropogenic noise. While much research has focused on traffic and urban noise, these are not the only anthropogenic noise sources humans have introduced to the environment. Wind farms are spreading rapidly worldwide (IEA, 2010) and as these are built away from urban areas, they may affect a different range of species. It is, therefore, especially important to know if and how bird songs are masked by wind turbine noise. Noise from wind turbines has most of its energy in the lower frequencies much like urban and traffic noise (Oerlemans et al., 2007), and therefore similar direct negative fitness effects can be expected. In addition, in Chapter 2 I have shown that wind turbine noise can affect bird communication.

Typically, territory defence is studied by simulating a territorial intrusion via playback of male song of the species of interest and recording several behavioural responses, such as latency to approach the speaker, closest approach to the speaker, number of flights, among others (e.g. Brindley, 1991; Brumm, 2004a; Poesel and Dabelsteen, 2005) or recording and analysing the songs sung in response to playback (Price et al., 2006; Geberzahn et al., 2009; Benedict et al., 2012). To investigate if and how territory defence is affected by anthropogenic noise, we simulated a territorial intrusion via playback in male robins (*Erithacus rubecula*) in the absence and presence of wind turbine noise.

Male robins respond to territorial intrusion by advertising ownership of the territory by song and calls, approaching the intruder (or speaker in a simulation),

threat posturing and finally attacking if the intruder does not retreat (Lack, 1965; Bremond, 1968). Song is thought to function predominantly in territorial disputes (Lack, 1965; Bremond, 1968) and as songs of robins can be masked by wind turbine noise (Chapter 2), robins are the ideal species to investigate if and how territorial intrusion is affected by wind turbine noise.

We aim to answer the following questions in this study: (1) do robins signal aggression by using low frequency elements in their song? (2) If so, how is the usage of these elements affected by wind turbine noise? (3) Do males respond differently in terms of non-singing behaviour (e.g. latency to approach signaller, closest approach and number of flights) to a territorial intrusion in the presence of wind turbine noise? To investigate these questions, we simulated territorial intrusion via playback in the absence and presence of wind turbine noise. As a negative control, we exposed male robins to wind turbine noise without simulating a territorial intrusion.

3.3 Material and methods

3.3.1 Construction of playback stimuli

We created three groups of playback stimuli: (i) robin songs; (ii) robin songs with wind turbine noise in the background; and (iii) wind turbine noise (Figure 10). To create the playback stimuli for (i) and (ii), we recorded songs of eight European robins that were singing at least 250m apart at Chopwell Woods near Newcastle upon Tyne in the UK (54º55'00''N, 1º47'25''W) on the 19th of April 2013 between 09.00 and 12.00 in the morning. We selected two minutes of these recordings from each robin for playback with an average song rate of 7.9 songs per minute (range = 6.5 – 9.5). In addition, to create playback stimuli for (ii) and (iii), wind farm noise was recorded on the 11th of March 2011 at a wind turbine of type REpower MM82 near Tow Law near Durham in the UK (54º45'35''N, 1º49'25''W). A section of one complete rotation of the turbine blades was isolated and repeated to make a two minute sound file, which was used for playback in group (iii). For playback stimuli group (ii), turbine noise was mixed with the files from group (i). All digital editing and construction of playback files was done in

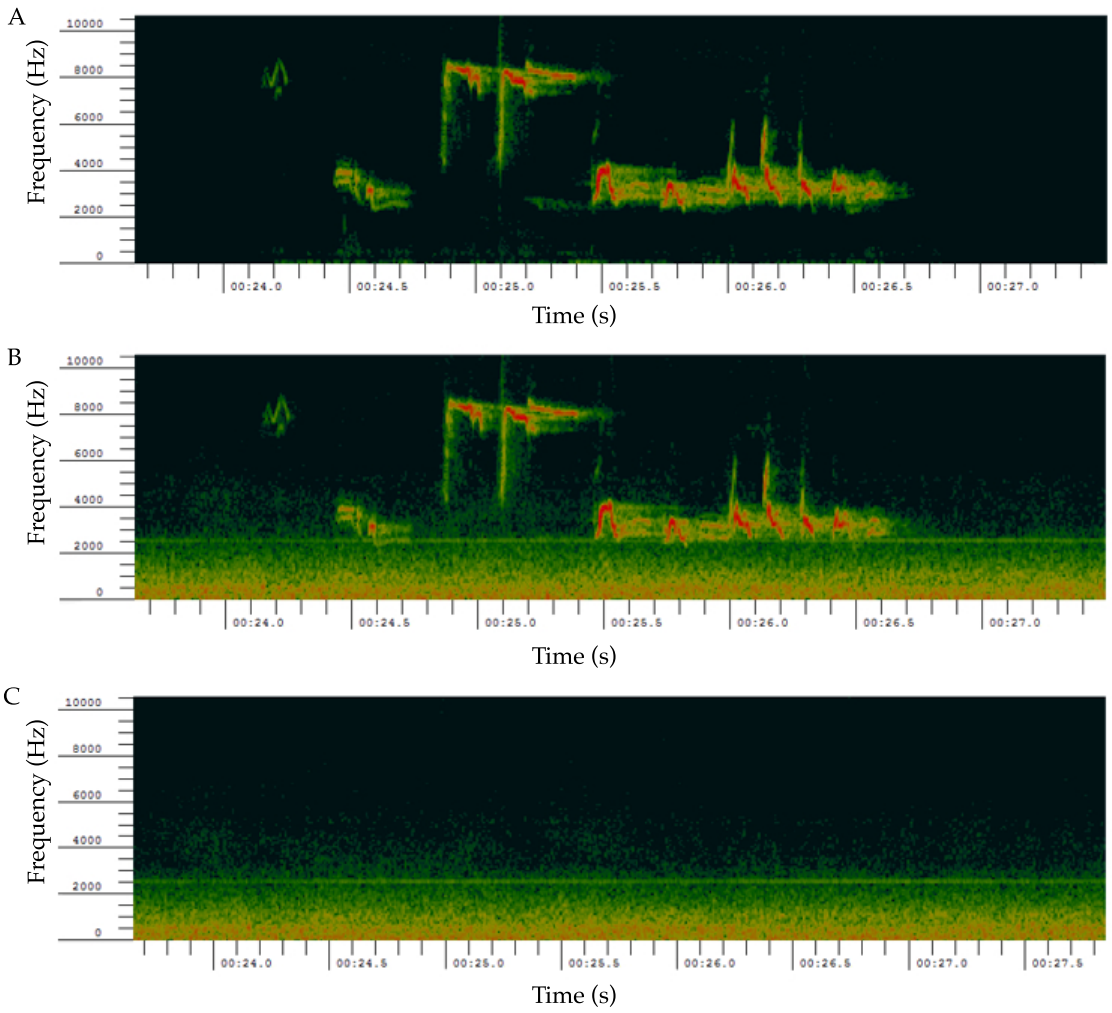

Figure 10: Spectrograms of playback stimuli. We simulated territorial intrusion via song playback. We played three different stimuli to individual male robins: (A) Robin song; (B) Robin song with wind turbine noise in the background; (C) Wind turbine noise. The playback lasted for two mins.

Audacity 1.3.13-beta. All songs were recorded with a Marantz PMD661 solid state recorder with a sampling rate of 44.1 kHz. The turbine noise was recorded with a Sennheiser ME-62 microphone and the songs were recorded with a Sennheiser ME-67 microphone.

3.3.2 Experiment procedure

Playback experiments were carried out at Chopwell woods near Newcastle upon Tyne in the UK (54º55′00″N, 1º47′25″W) from the 20th to 23rd of April 2013 between 06.00 and 11.00 in the morning. This site was selected because it is away from busy roads, wind farms and other anthropogenic noise sources. The

playback experiments were carried out in this brief period to ensure that all males tested were in the same breeding stage. In addition, this period was chosen as deciduous trees are still without foliage which ensured good observation and tracking of the behaviour of the robins. Each playback experiment lasted six minutes: two minutes of pre-playback recording and observation, two minutes playback recording and observation and two minutes of post-playback recording and observation. Twenty-four different male robins were tested in three different groups (eight males per group): (i) robin songs; (ii) robins songs with wind turbine noise; (iii) wind turbine noise (see Figure 11). The third group acted as a negative control, to ensure that any behaviour recorded from the birds was not due to our presence or the speaker. In addition, noise exposure could lead to the robin moving away (McLaughlin and Kunc, 2013) and thus if we recorded this in group (ii), we would also have to record this in group (iii) if this behaviour was due to wind turbine noise. The amplitude levels of the playback were 82 dB SPL for robin songs measured at 1m from the loudspeaker with a SL-8850 sound level meter (Castle Group Ltd) and 70 dB SPL at 1m for the wind turbine noise. Before playback began, we located two song posts of the singing male and positioned the speaker, Skytec ST-100, between the two song posts towards the targeted bird. Playback was only initiated if the targeted bird was singing. To ensure that a male was not tested twice or received a playback with its own song or that of its neighbour, we recorded their location on a map and ensured sufficient distance between them (minimum distance between playback and subject: 340m). In addition, we did not test neighbouring robins on the same day (minimum distance between subsequent subjects: 210m). We randomised the order of the playback via a random number generator to ensure that a specific treatment was not favoured at a particular day or time. The initial distance between the speaker and the targeted bird measured with a Leupold RX-600i digital laser rangefinder did not differ between the three groups (One-way ANOVA: $F(2,20)=0.812$, $p=0.458$). We recorded the following non-singing behavioural variables: latency to approach (that is time in seconds to first flight after playback started), the closest approach during playback, the number of flights and the number of full songs (as opposed to the shorter, quieter subsong). We counted the number of flights and the number of full songs

two minutes (2 min) before, 2 min during and 2 min after playback. We also noted if the robin male responded by threat posturing. In addition, we recorded their vocalisations during the same periods with a Sennheiser ME-67 microphone on a Marantz PMD661 solid state recorder with a sampling rate of 44.1 kHz and an accuracy of 16-bit. Using the same equipment, a spoken commentary of the response variables was recorded during the experiment.

3.3.3 Acoustic analyses

In total, we recorded 911 songs from 24 robins (mean = 38 per male, range = 16-60) during the experiments. Songs that overlapped with songs from the playback or overlapped with songs from other birds in the background were not analysed as it would not be possible to take reliable measurements from these songs. Therefore, in total, we analysed 839 songs. The elements of each of these songs were automatically detected via the automatic parameter measurement function in Avisoft SASLab Lite 5.2.07 (Figure 12). Due to the lower-pitched part of the songs being potentially masked by noise, reliable measurements of minimum frequency is impeded (Zollinger et al., 2012) and thus we measured the peak frequency of each detected element. In total, we analysed 5057 elements from group (i) (mean = 632 per male, range = 290-1143), 4451 elements from group (ii) (mean = 556 per male, range = 187-732) and 4485 elements from group (iii) (mean = 561 per male, range = 268-852).

3.3.4 Statistical analyses

For each robin, we calculated the frequency distribution of the song elements for the duration of the 2 min before, 2 min during and 2 min after playback. For each of these periods, we counted the number of elements that had its peak frequency in a particular 100 Hz band and divided this number by the total number of elements that the robin had sung during these two minutes. We chose the frequency range to be from 1800 Hz to 9900 Hz as these were the minimum and maximum peak frequency of the elements measured between all the robins.

If robins are changing their frequency distribution due to masking effects of

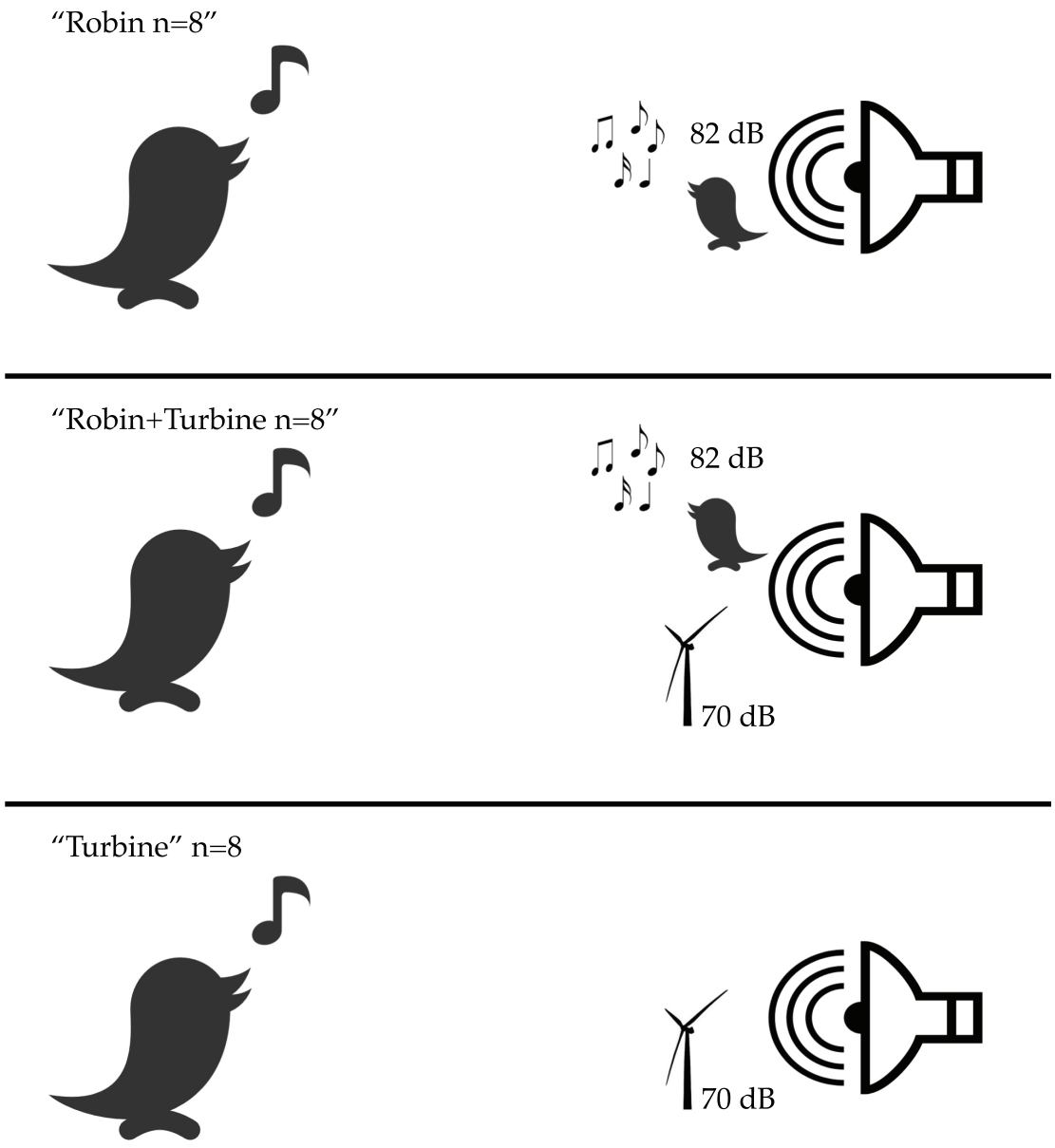

Figure 11: Overview of the experimental setup. We simulated territorial intrusion via song playback. We played three different stimuli to individual male robins: 1) Robin song; 2) Robin song with wind turbine noise in the background; 3) Wind turbine noise. The amplitude levels of the playback were 82 dB SPL for robin songs measured at 1m from the loudspeaker and 70 dB SPL at 1m for the wind turbine noise. The playback lasted for two mins. The robins were observed for two minutes (2 min) before, 2 min during and 2 min after playback. The following variables were recorded: number of flights, number of songs, latency to approach the speaker and closest approach to the speaker. In addition, we recorded the vocalisations of the robins during the experiment for acoustic analyses.

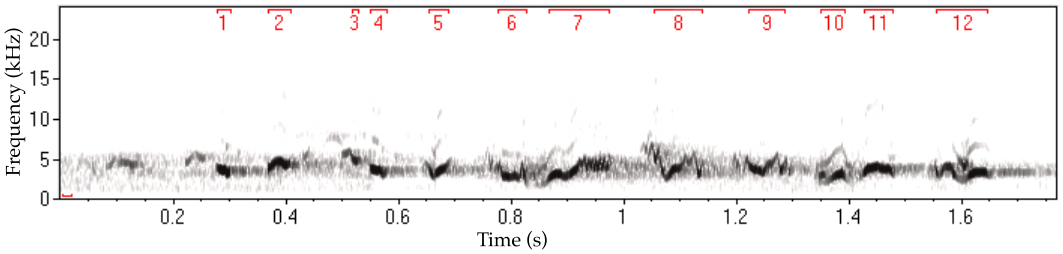

Figure 12: Automated detection of song elements via the automated parameter measurement function in Avisoft SASLab. Detected song elements are shown with a red bar above the element. In this example there are 12 detected song elements. For each detected element we measured the peak frequency, which is the frequency at the maximum amplitude.

turbine noise, this is most likely to occur in the lower frequency range as this is the range where most turbine noise energy resides. We therefore chose to focus on the frequency range which contained 20% of the lowest elements that robins used before playback, which was between 1800 Hz to 3300 Hz. As there was no difference between treatments for robins before playback, we pooled the data for this period for all 24 robins together. For each group we then compared the frequency distribution of song elements during playback or after playback with the frequency distribution before playback via Wilcoxon rank sum tests, as data could not be transformed to normality. All p-values were adjusted via the sequential Bonferroni correction or otherwise known as the Holm's correction (Holm, 1979), as this method is a more powerful procedure than the conventional Bonferroni correction (Goldstein, 2011).

3.4 Results

3.4.1 Non-singing behaviour

Males that were tested with robin songs with wind turbine noise performed more flights during playback than males that were tested with robin songs alone (groups median: 6 flights vs 3.5 flights respectively), but this effect was not statistically significant (Wilcoxon rank sum test: W=16.5, N=16, p=0.111). Only two of the eight robins used threat posturing when territorial intrusion was simulated with background turbine noise while without noise four of the eight

robins used threat posturing. We found no difference between the two groups in the number of flights performed after playback (Wilcoxon rank sum test: W=25.5, N=16, p=0.514), their closest approach during playback (Two sample t-test: t=1.160, df = 13.49, p=0.266) or their latency to approach the speaker (Wilcoxon rank sum test: W=25, N=16, p=0.493) (Figure 13). In addition, the number of songs performed during or after playback did not differ between the two groups (Two sample t-test during: t=0.100, df=13.995, p=0.922; after: t=0.392, df=13.913, p=0.701).

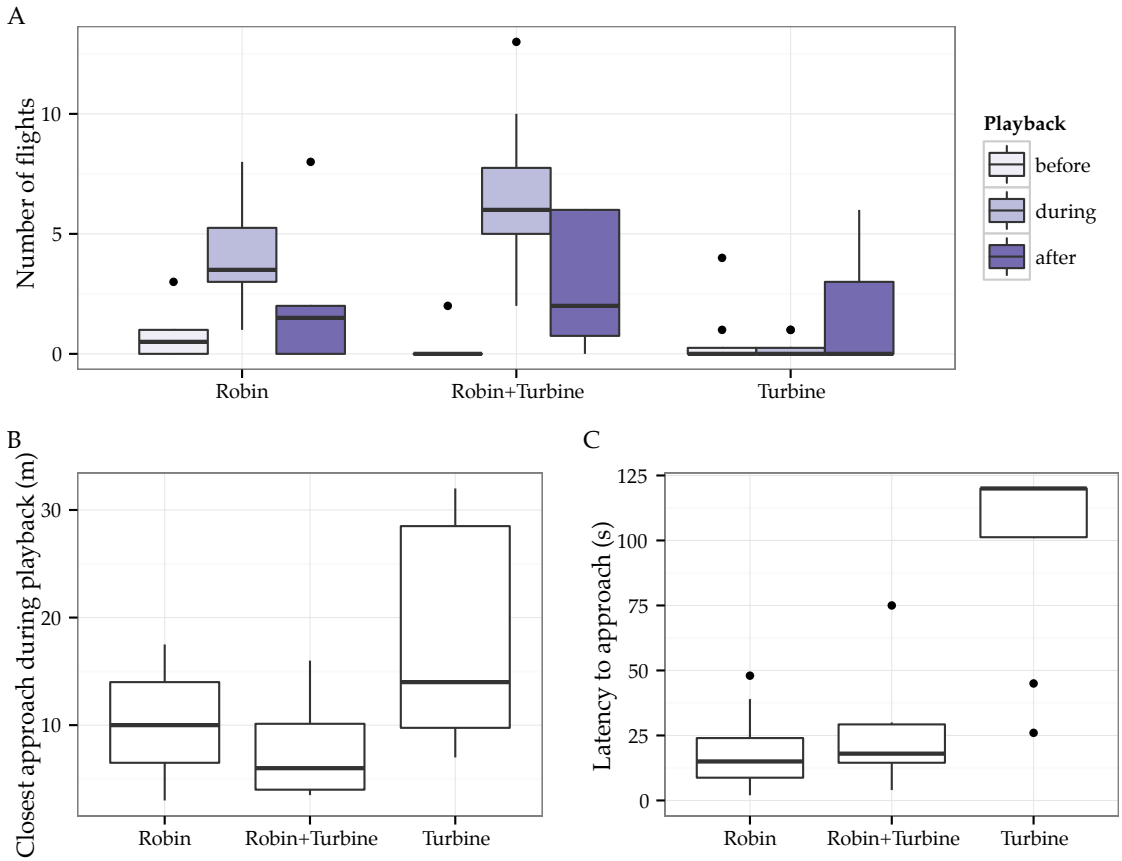

Figure 13: Responses of European robins to playback of three different stimuli. The stimuli were: Robin song ("Robin"), robin song with wind turbine noise in the background ("Robin+Turbine"), wind turbine noise ("Turbine"). The three different groups are displayed on the x axis. Boxplots for the three different responses recorded: (A) number of flights performed before, during or after playback; (B) closest approach of the robin during playback; (C) latency to approach the speaker during playback (that is the first flight towards the speaker after playback commenced).

Males that were exposed to wind turbine noise alone did not respond to the playback as there was no difference between the number of flights performed

before and during playback (Wilcoxon rank sum test: W=33, N=16, p=0.945) or between before and after playback (Wilcoxon rank sum test: W=20.5, N=16, p=0.627). In addition, robin males did not approach the speaker in response to wind turbine noise alone, with no difference between their initial distance to the speaker and their closest approach to the speaker (Wilcoxon rank sum test: W=19.5, N=16, p=0.560). Furthermore, the number of songs performed by each robin in this group either during or after playback did not differ with the number of songs sung before playback commenced (two sample t-test before vs during: t=0.255, df=13.658, p=0.802; before vs after: t=1.268, df=11.989, p=0.229).

3.4.2 Frequency distribution of song elements

We found that robins that were tested with robin songs with wind turbine noise in the background sang only 12% of their song elements within the range of 1800 to 3300 Hz during playback as opposed to 20% before playback commenced, which is a significant 40% reduction (Wilcoxon rank sum test: W=22584.5, N_{before}=24, N_{during}=7, p=0.004) (Figure 14b). No change in element usage in this frequency range was observed during playback in robins that were tested with robin songs alone (Wilcoxon rank sum test: W=18409.5, N_{before}=24, N_{during}=8, p=0.675) or with only turbine noise alone (Wilcoxon rank sum test: W=22891.5, N_{before}=24, N_{during}=8, p=0.614) (Figure 14a, c). After playback, robins exposed to robin songs alone sang 31% of their song elements within the range of 1800 to 3300 Hz, which is a significant increase of 55% compared to before playback commenced (Wilcoxon rank sum test: W=5318.5, N_{before}=24, N_{after}=8, p=<0.001). There was no difference in the frequency distribution of the song elements after playback compared with before playback in the other two groups (Wilcoxon rank sum tests: Group Robin+Turbine W=18132.5, N_{before}=24, N_{after}=7, p=0.513; Group Turbine W=18290.5, N_{before}=24, N_{after}=6, p=0.141) (Figure 14d, e, f). All p-values were adjusted using sequential Bonferroni correction.

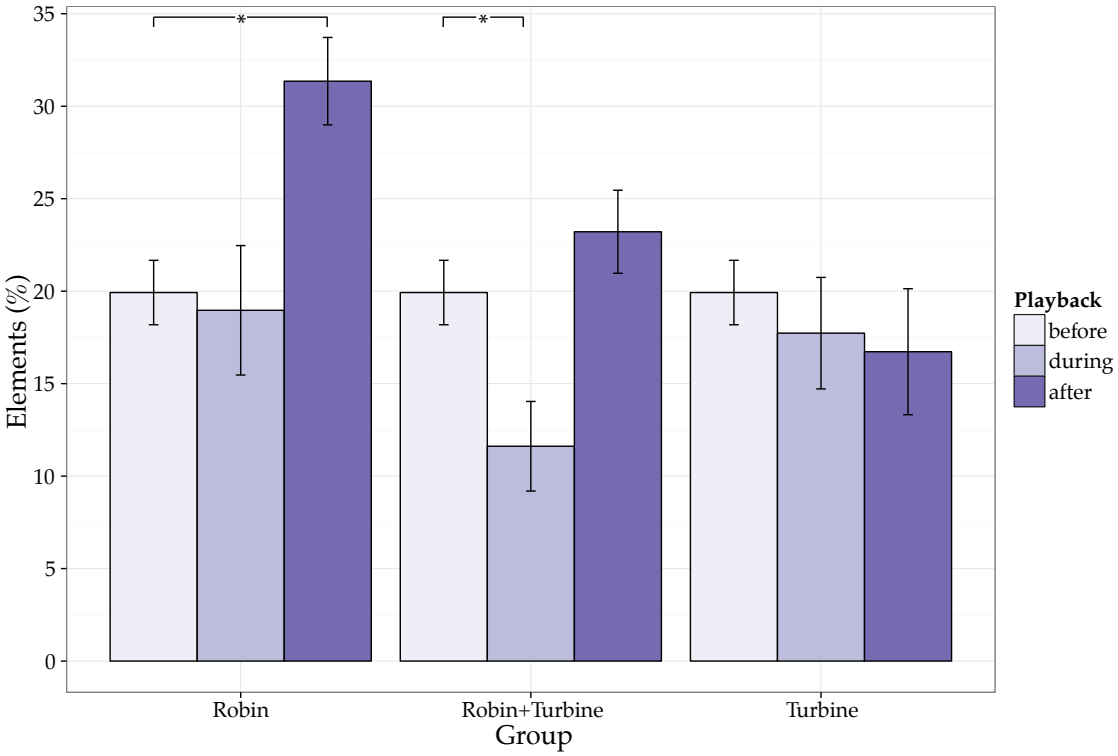

Figure 14: Distribution of peak frequency of elements in robin songs between 1800 and 3300 Hz. The robins were exposed to robin songs (left), robin songs with turbine noise (middle) or wind turbine noise (right). The bars show mean peak frequency from the eight robins in each group and standard errors. The asterisks denote significant difference.

3.5 Discussion

We found that robin males increased their usage of low frequency elements in the range 1800 Hz to 3300 Hz by 55% when territorial intrusion was simulated. Interestingly, in the presence of wind turbine noise, we found that males decreased their use of these elements by 40% compared to before playback. The singing behaviour of robins was not altered when we exposed males to turbine noise without territorial intrusion. In the non-singing behaviour response parameters measured, male robins' territorial behaviour was not altered significantly by wind turbine noise when territorial intrusion was simulated. In addition, birds exposed to wind turbine noise alone were unaffected by the noise as none of the behaviours we measured changed in response to the playback.

3.5.1 Robins signal aggression by using low frequency elements

The 55% increase in usage of low frequency elements when challenged indicates that robins signal aggression in this manner. This supports the motivation-structural hypothesis, which suggests that low frequency signals are used in hostile encounters as they can indicate the size of the animal producing the sound because frequency and body size are correlated (Morton, 1977). Therefore, information on body size could indicate the sender's fighting ability. A similar result to ours has been reported for canyon wrens (*Catherpes mexicanus*), which lowered the low frequencies of their songs when territorial intrusion was simulated (Benedict et al., 2012).

3.5.2 Wind turbine noise affects the usage of low frequency elements during territorial intrusion

In contrast, in the presence of wind turbine noise, robins decrease their usage of low frequency elements by 40% during territorial intrusion. This indicates that males responded with a much reduced aggressive signal in the presence of wind turbine noise and therefore might not be able to deter rivals successfully. They may have chosen signal detection over signal efficacy due to the masking effect of wind turbine noise. This frequency shift could be related to the Lombard effect (Lombard, 1911): as higher frequency elements can be produced more easily at higher amplitudes, birds may switch to these frequencies to increase signal detection (Nemeth et al., 2013). In addition, it is much more beneficial for the signaller to increase the amplitude of the signal than to increase the minimum frequency. This is because the maximum communication distance of a signal in noise is much more affected by an increase in amplitude than an increase in minimum frequency (Nemeth and Brumm, 2010; Parris and McCarthy, 2013).

Male robins may have switched to other strategies to signal aggression to the intruder. Robins use threat posturing where they show their red breast to the intruder. However, we found that only 25% of the males tested used threat posturing in the presence of wind turbine noise, while in the absence of noise 50% of the males used threat posturing. This indicates that in presence of turbine

noise, robins have not switched to visual cues to signal aggression.

3.5.3 No frequency shift when no territorial intrusion is simulated

Interestingly, we did not find a frequency shift in our control group when we exposed robins to wind turbine noise but did not simulate territorial intrusion. The closest approach to the speaker was significantly further away for the control group than for the birds in the groups where we simulated territorial intrusion (median 14m in group Turbine, median 6m in group Robin+Turbine). Therefore, for the control group the noise levels might not have been sufficient enough to affect their singing behaviour. Robins have been found to increase the minimum frequency of their song in response to artificially created urban noise (Montague et al., 2012; McLaughlin and Kunc, 2013). In these studies, the robin was either attracted to the speaker by playing robin song (and therefore simulating territorial intrusion), or was exposed to much higher levels of noise (90 dB at 1m from the speaker vs 70 dB in this study) when it was between 7m and 15m away from the speaker. This difference in experimental protocol might explain why our results differ to those from previous studies. In addition, we have undertaken a more extensive analysis of robin song structure by measuring the distribution of elements in songs which gives a better overview of frequency usage (Nemeth et al., 2013).

3.5.4 Non-singing behaviour not affected

For the non-singing behaviour parameters measured, we did not find any significant differences between the groups. This may be due to the fact that the auditory signal is enough to deter a rival without the need for visual cues as song is thought to be the main driver in territorial disputes (Lack, 1965; Bremond, 1968). However, we did find that in the presence of wind turbine noise fewer males used threat posturing, which is a further indication that these males responded less aggressively to the simulated intruder, but due to a low sample size we were not able to test this statistically. Further research is needed to investigate how threat posturing is used in the presence of anthropogenic noise

and whether robins use additional methods to signal aggression which could be used in noisy environments.

3.5.5 Conclusion

European robins use low frequency elements to signal aggression and may therefore win territorial disputes while avoiding fights. In the presence of wind turbine noise, European robins substantially reduced their usage of low frequency elements of their song during simulated intrusion, which indicates that they responded with a much reduced aggressive signal. They may have sacrificed signal efficacy for signal detection due to the masking effects of wind turbine noise. As a result, it may affect their ability to deter a rival, which may lead to expenditure of extra time and energy and could lead to increase risks of injury and as a consequence could cause reduced breeding success. The effect that anthropogenic noise has on territory defence may be one of the important causes of the reduced breeding success found near noisy areas, including wind farm areas. Since wind farms are built in relatively quiet areas, this effect might be important for planning wind farms.

To our knowledge, this is the first study that shows that territory defence is affected by anthropogenic noise. Further research is needed to investigate what consequences singing songs with substantially fewer low frequency elements has. For example, it would be interesting to study how males respond to intruders which use low frequency elements vs intruders that do not use low frequency elements. In addition, more research is required to study whether other sources of anthropogenic noise have similar effects on territorial intrusion. However, similar effects can be expected as wind turbine noise has most of its energy in the lower frequencies like urban and traffic noise. It is likely that other species that mainly use song in their territorial disputes are also affected, although further research is needed to investigate this.

Chapter 4. The use of automated bioacoustic recorders to replace human wildlife surveys: an example using nightjars

4.1 Related manuscript

Zwart, M. C., Baker, A., McGowan, P. J. K., Whittingham, M. J. (2014). The use of automated bioacoustic recorders to replace human wildlife surveys: an example using nightjars. *PLoS ONE*, **9**(7):e102770.

4.2 Abstract

To be able to monitor and protect endangered species, we need accurate information on their numbers and where they live. Survey methods using automated bioacoustic recorders offer significant promise, especially for species whose behaviour or ecology reduces their detectability during traditional surveys, such as the European nightjar. In this study we examined the utility of automated bioacoustic recorders and the associated classification software as a way to survey for wildlife, using the nightjar as an example. We compared traditional human surveys with results obtained from bioacoustic recorders. When we compared these two methods using the recordings made at the same time as the human surveys, we found that recorders were better at detecting nightjars. However, in practice fieldworkers are likely to deploy recorders for extended periods to make best use of them. Our comparison of this practical approach with human surveys revealed that recorders were significantly better at detecting nightjars than human surveyors: recorders detected nightjars during 19 of 22 survey periods, while surveyors detected nightjars on only six of these occasions. In addition, there was no correlation between the amount of vocalisation captured by the acoustic recorders and the abundance of nightjars as recorded by human surveyors. The data obtained from the recorders revealed

that nightjars were most active just before dawn and just after dusk, and least active during the middle of the night. As a result, we found that recording at both dusk and dawn or only at dawn would give reasonably high levels of detection while significantly reducing recording time, preserving battery life. Our analyses suggest that automated bioacoustic recorders could increase the detection of other species, particularly those that are known to be difficult to detect using traditional survey methods. The accuracy of detection is especially important when the data are used to inform conservation.

4.3 Introduction

Information on where species occur and in what numbers are important for an increasing variety of reasons, whether it is for understanding environmental change, assessing and monitoring conservation status or as part of a legislative or policy process. A variety of approaches, sampling designs and field protocols have been developed to meet such needs, such as point counts, transect counts and mapping (Bibby et al., 1992; Krebs, 1999; Southwood and Henderson, 2000; Sutherland et al., 2004a). The increasing demand for these data, in particular for informing environmental impact assessments (EIAs), means that new opportunities should be explored to align technical developments with sound sampling design and appropriate field protocols. This is critical if cost-effective and accurate ways of providing data on species occurrence and abundance are to be found. This need is most evident for species that are considered threatened and which are often described as cryptic because their behaviour or ecology substantially reduces their detectability during standard surveys (Joint Nature Conservation Committee, 2004). The European nightjar (*Caprimulgus europaeus*), which is listed on the Annex I Birds Directive (Directive 2009/147/EC, 2010) and is a Biodiversity Action Plan Priority Species in the UK, presents precisely these challenges. In particular, nightjars are difficult to detect as they are nocturnal and very well camouflaged. The traditional way to survey for them is via surveys at either dusk or dawn. Their vocalisation is less complex than many other bird songs, consisting primarily of a series of pulses, and thus lends itself to automated classification (Brandes, 2008). We therefore explored whether

automated acoustic recorders offer a cost-effective way of increasing the reliability of surveys when compared with current approaches.

Audio recording has already been used as a replacement for point counts (Rempel et al., 2005; Swiston and Mennill, 2009; Celis-Murillo et al., 2012; Campbell and Francis, 2012) or for identifying individuals (Gilbert et al., 1994; Peake and Mcgregor, 2001). However, in the case of point counts, the potential for recorders to be deployed for longer periods has not been investigated. In addition, as computer technology has advanced and faster computers have become more readily available there is potential for using automatic call recognition for the audio recordings collected, where the species that are on the audio recordings are automatically classified based on an algorithm. Automated classification has been successfully applied to detect a range of species including woodpecker spp. (Swiston and Mennill, 2009) and antbird spp. (Trifa et al., 2008).

In this study we examined the utility of automated bioacoustic recorders and the associated classifying software as a way to survey for wildlife, using the nightjar as an example. We compared detection by traditional surveys, described by Gilbert et al. (1998), with results obtained using bioacoustic recorders. We end by making recommendations for the best use of these recording devices for nightjars and other species.

4.4 Materials and methods

4.4.1 Ethics statement

Permission to carry out the fieldwork at these sites was obtained from the Forestry Commission under permit number 144/12. As this was a purely observational study, no specific ethical approval was needed.

4.4.2 Study sites

We chose two Forestry Commission sites in Northumberland, UK, where nightjars were previously recorded: Slaley Forest (geographic coordinates of the central point of the site: 54°53′11″N, 2°5′21″W) and Fourlaws (geographic coordinates

of the central point of the site: 55°8′50″N, 2°7′11″W). These sites comprised a mixture of different age stands of coniferous woodland, heather moorland, and a small amount of deciduous woodland around the edges of the conifer stands.

4.4.3 Traditional site surveys

Between mid-June and the end of July 2012, which is during the breeding season of the nightjar (Gilbert et al., 1998), we performed four surveys at each site. Each survey was separated by a two-week interval. Surveys began at dusk on clear nights with low to no wind. Survey methods followed the methods described in Gilbert et al. (1998). In short, human surveyors walked a route of 6 km at a speed of 3-4 km/h and stopped every few minutes to listen for churring or calling nightjars. The starting point and route of surveys were randomised so that we did not favour a particular area at the same time. When a nightjar was located, its locality and behaviour was recorded with a handheld GPS device (Trimble Juno 3B) which ran ArcPad 10.0.

4.4.4 Recorders

At each site three full spectrum recorders (SM2+, Wildlife Acoustics Inc.) were placed, one per km^2 (six in total), during the same period as the traditional site surveys (see Figure S3). The recorders had a microphone on both the left and right side and were attached to a tree. The recorders were set to record throughout the night between 22.00 and 04.30 the following morning on both channels with a gain of +48dB and sampling rate of 44100Hz (see Table S1 & S2 for details). Recordings were saved to disk at 30-minute intervals to prevent the loss of an entire evening's recordings in the event the batteries ran out. These 30 min. segments were saved in a compressed (native .wac) format. Each recorder continued to record until the batteries ran out of power (SM2+ with GPS average 14.1 hours (range 13.5-15), SM2+ without GPS average 25.4 hours (range 23.5-26.5)). This resulted in an average recording period of four nights (range three to five nights). We used Energizer D 1.2V NiMH 2500mAh rechargeable batteries. These batteries were replaced every week for nine weeks between mid-June and mid-August 2012. The traditional surveys took place at the beginning of week 1, 3, 5 and 7. Recorder

6 was put out at week 4 and recorders 4-6 were taken down after week 7 due to site permission restrictions. All other recorders were active for the nine week duration.

4.4.5 Recognizer

We used an automated recognizer rather than listening to 1948 hours of recordings (196, 198, 194.5, 142, 145.5 and 98 hours of recordings per channel on recorder 1-6 respectively). Recognizers for churring and for flight calls were generated in Song Scope 4.1.3A (Wildlife Acoustics Inc.). A fast Fourier Transformation (FFT) size of 64 and 50% overlap was used with a frequency range of 500-3000Hz. These settings were chosen to get a good temporal resolution of the individual pulses that make up the churring of the nightjar. In total 181 suitable sections of recorded nightjar churring were selected from 12 files from two different recorders and loaded in the program as training data for the churring recognizer. The 12 files corresponded to 3.8% of the total number of files where some nightjar churring was detected. On average the sections were of 3.67 $\pm$ 2.16 (1SD) seconds in length. For the flight call recognizer 32 flight calls were selected from 15 files from four different recorders. These files were 5.5% of the total number of files where some nightjar flight calling was detected. The sections that were selected were on average of 0.40 $\pm$ 0.11 (1SD) seconds in length. When building the recognizer we consulted with Wildlife Acoustics Inc. in order to choose the settings that achieved the best results. Several different configurations of settings were tested on a small sample set of the data (see Table S3 for details on the different configurations tested). This sample set included both positive and negative controls: that is, audio files where it was known that nightjars were churring and flight calling, and also audio files where it was known that no nightjars were churring or flight calling. We chose the final configuration settings in order to achieve the lowest false positive and false negative rates. This criterion resulted in the following settings for building the recognizer for churring: maximum complexity: 16, maximum resolution: 15, sample rate: 8kHz, FFT size: 64, FFT overlap: 1/2, frequency minimum: 8, frequency maximum: 18, background filter: 1s, maximum syllable length: 148ms,

maximum syllable gap: 148ms, maximum song length: 5980ms, dynamic range: 10 and algorithm: 2.0. The following settings were used for the flight call recognizer: maximum complexity: 16, maximum resolution: 10, sample rate: 8kHz, FFT size: 256, FFT overlap: 1/2, frequency minimum: 28, frequency maximum: 88, background filter: 1s, maximum syllable length: 256ms, maximum syllable gap: 0ms, maximum song length: 352ms, dynamic range: 20, algorithm: 2.0.

4.4.6 Analysis

Extracting data from the recorders

For each compressed wac file the two channels were saved as separate files and converted into a wav file with wac2wav 3.3.0 (Wildlife Acoustics Inc.). A batch process was set up where the nightjar activity on the wav files from both channels were classified by Song Scope using the recognizer. Before running the batch process we chose to save only the results that had a score above 40.0% and a quality above 0. The score value is on a scale from 0.00% to 100.0% and represents the statistical fit of the candidate nightjar vocalisation to the nightjar recognizer model. The quality value is on a scale from 0.00 to 99.99, and reflects how well a set of secondary parameters of the candidate nightjar vocalisation match with the training data used to build the recognizer. We first did a batch process with minimum score of 0% and minimum quality of 0 on three audio files known to have a substantial amount of nightjar vocalisations and ordered the results by score. We then looked through these results and noted the minimum score for audio files that included genuine nightjar vocalisations, which in our case was 40.0%. This score value was then used in the batch process which was run on all files. In consultation with Wildlife Acoustics Inc., we decided to keep the quality value at 0 to reduce false negatives. We listened to 20 hours of recordings (1% of the total) to estimate false negatives with these settings. The churring recognizer had a false negative rate of 4.4%, while the flight call recognizer had a false negative rate of 13.4%. In our 20-hour sample, false negatives were caused exclusively by either failing to recognize very distant nightjar vocalisations or, in the case of the flight calls, by simultaneous churring overlapping and obscuring

the calls. Finally, all files in our sample where the recognizer did not detect a nightjar at all were verified as true negatives in our listening experiments. Thus, the recorders performed very well at detecting nightjar presence, with any false negatives restricted to the recognition of individual calls.

After the batch process finished, all positive results were manually verified, and false positives were deleted from the analysis (in total we found 766,275 false positives across all recorders). To understand the cause of these false positives, we manually investigated all cases where the recognizer reported more than 100 results in one 30-min file but where no churring or flight calls had been recognized (80.1% of all false positives). This included 84.3% of the false positives for the churring analysis, of which 99.2% were due to bad weather (heavy wind or rain), and 69.9% of the false positives for the flight call analysis, of which 88.7% due to bad weather (heavy wind or rain). False positives could be reduced by excluding recordings when there is bad weather. In addition, during a batch process, settings for score and/or quality could be increased to reduce the false positives (e.g. increasing the score value to 65.0% reduced our total false positives to 53,789) but this has a tradeoff with the false negatives as these will then be increased. The results from the churring recognizer were split into individual 60 second (s) sections. If any registration of a churring nightjar was made (of any duration) during a 60s sample then that sample was recorded as a positive registration. For each 30 minutes of recording we listed the total number of positive 60s samples. The results from the flight call recognizer were counted to give the total number of flight calls recognized in each 30 minutes segment of recording. The results were loaded in R 3.0.0 (R Core Team, 2013) for further analysis.

Comparison of traditional surveys with recorders

Data from traditional surveys were plotted in ArcGIS. We then determined the nearest recorder within a 500m radius for each located nightjar registration and for each recorder, and each visit, we counted the number of nearest registrations. We transformed these data and the data from the recorders into presence/absence data. We compared the presence/absence data from the traditional surveys with the presence/absence data from the recorders with McNemar tests: that is, if one

or more nightjars were located nearest to recorder A on visit A it was scored as a '1'; similarly if the audio recorders yielded data (either from the churring or the flight call analysis) for one or more nightjars, over a particular time period (see below), then it was scored as a '1'. The survey data were compared with the audio data that were obtained during weeks 1, 3, 5, and 7 (n = 22; we had four visits and six recorders. As recorder 6 was not deployed until week 4, the data from visit 1 and 2 near the subsequent location of this recorder could not be compared with any audio data).

Furthermore, we tested whether the survey data on the abundance of nightjars at each recorder was correlated with the amount of vocalisation that was recorded on the same recorder. A correlation would be expected if traditional human surveys were accurate and the amount of vocalisation could then be used to infer numbers of nightjars. For this, we fitted two generalised linear mixed models (GLMMs) with a Poisson error structure, with recorder as a random effect, and the amount of vocalisation (either churring or flight calls) as fixed effects. We compared the survey data with recordings made during the week after the survey (n = 22). For these tests the data were also split per recorder, per visit.

Based on the recorded data, when is it best to survey for nightjars?

To test what time during the night nightjars are most active, we split the data into three time periods: 22.00-00.00 (dusk), 00.00-02.30 (middle), 02.30-04.30 (dawn). We then compared the activity (churring or flight calls) recorded via a GLMM with a Poisson error structure. As the activity of nightjars differed between recorders, we fitted the different recorders as a random effect. Since the total recording time was different between recorders, we fitted this as an offset.

Comparison of recorder settings

We subsampled the data collected by the recorder to inform how to use the recorders. We were particularly interested in testing if the recorder could be set to record for shorter time periods while achieving similar results; this would save battery power so that the recorders could be left out for longer periods.

Therefore, we compared the following settings that the recorder might be set to: (1) recording 10 mins every hour, (2) only dusk (22.00-00.00), (3) only the middle of the night (00.00-02.30) (4) only dawn (02.30-04.30), (5) dusk and dawn, and (6) recording all night. For this we selected the respective data from the data collected via the recognizers. For example (1) for every hour of recording the first 10 mins was selected (e.g. 22.00-22.10, 23.00-23.10) and the total number of churring mins and flight calls was calculated. We then calculated the total number of nights "nightjar_nights" that a nightjar was detected by each recorder (either via the churring or the flight call recognizer). We scored each night that a nightjar was detected as '1' (nights with no nightjars were scored as '0'). We then fitted a generalised linear model with a Poisson error structure with the formula: nightjar_nights = setting + offset(log(total_nights)), where the "total_nights" is the total number of nights that the recorder was on.

All GLMMs were fitted with the package lme4 (Bates et al., 2013). Pairwise post-hoc analysis was done via the glht function in the multcomp package (Hothorn et al., 2008). We used an alpha value of 0.05 to assess the significance of results. All statistical tests were performed in R version 3.0.0 (R Core Team, 2013).

4.5 Results

4.5.1 Comparison of traditional surveys with recorders

We compared the detection of nightjars via traditional human surveys with detection by bioacoustic recorders using the data recorded at the same time as the human surveys (22:00-00:00 on survey nights), and found that the recorders detected nightjars during five of the eleven survey periods, while the humans only detected nightjars during three of the eleven survey periods. When we looked at nightjar detection by the recorders during the whole survey night (22:00-04:30), we found that detection by the recorders increased to eight of the eleven survey periods. Moreover, when we compared the results of the human surveys with recordings made during the whole week of the survey, we found that the recorders detected nightjars during 19 of 22 survey periods, while surveyors detected nightjars on only six of these occasions; human surveyors

never detected a nightjar that the recorders failed to detect. This equates to a 217% increase in detection of nightjars using bioacoustic devices when compared with human surveyors.

We found no correlation between the abundance of nightjars found by the traditional surveys at each recorder and the amount of vocalisation (either churring activity or flight calling activity) that was recorded on the same recorder (churring activity: estimate = 0.000, std error = 0.004, z-value = 0.087, p-value = 0.931; flight calling activity: estimate = -0.025, std error = 0.023, z-value = -1.093, p value = 0.274). For example, at recorder 1 there was a substantial amount of nightjar activity during dusk and dawn (see Figure 15), but nightjars were not detected during any of the four traditional human surveys. We therefore judge it unwise to provide accurate estimates of nightjar numbers in our study area. However, we can give a crude estimate based on the human surveys via methodology described by Gilbert et al. (1998). This yielded a count of eight different churring males, but it is very likely to be an underestimate.

4.5.2 Based on the recorded data, when is it best to survey for nightjars?

We found that nightjars are most active at dawn and least active during the middle of the night based on their churring activity (see Figure 15a). Churring activity during dawn was significantly greater than during dusk or middle of the night (dusk-dawn: estimate = -0.531, std error = 0.048, z-value = -11.172, $p<0.001$; middle of night-dawn: estimate = -1.018, std error = 0.052, z-value = -19.729, $p<0.001$). There was more churring activity during dusk than during the middle of the night (estimate = -0.487, std error = 0.054, z-value = -9.075, $p<0.001$). We found no significant difference in flight calling activity during the three periods (dusk-dawn: estimate = -0.009, std error = 0.053, z-value = -0.169, p = 0.984; middle of night-dawn: estimate = 0.003, std error = 0.051, z-value = 0.053, p = 0.998; middle of night-dusk: estimate = 0.012, std error = 0.046, z-value = 0.253, p = 0.965; see Figure 15b).

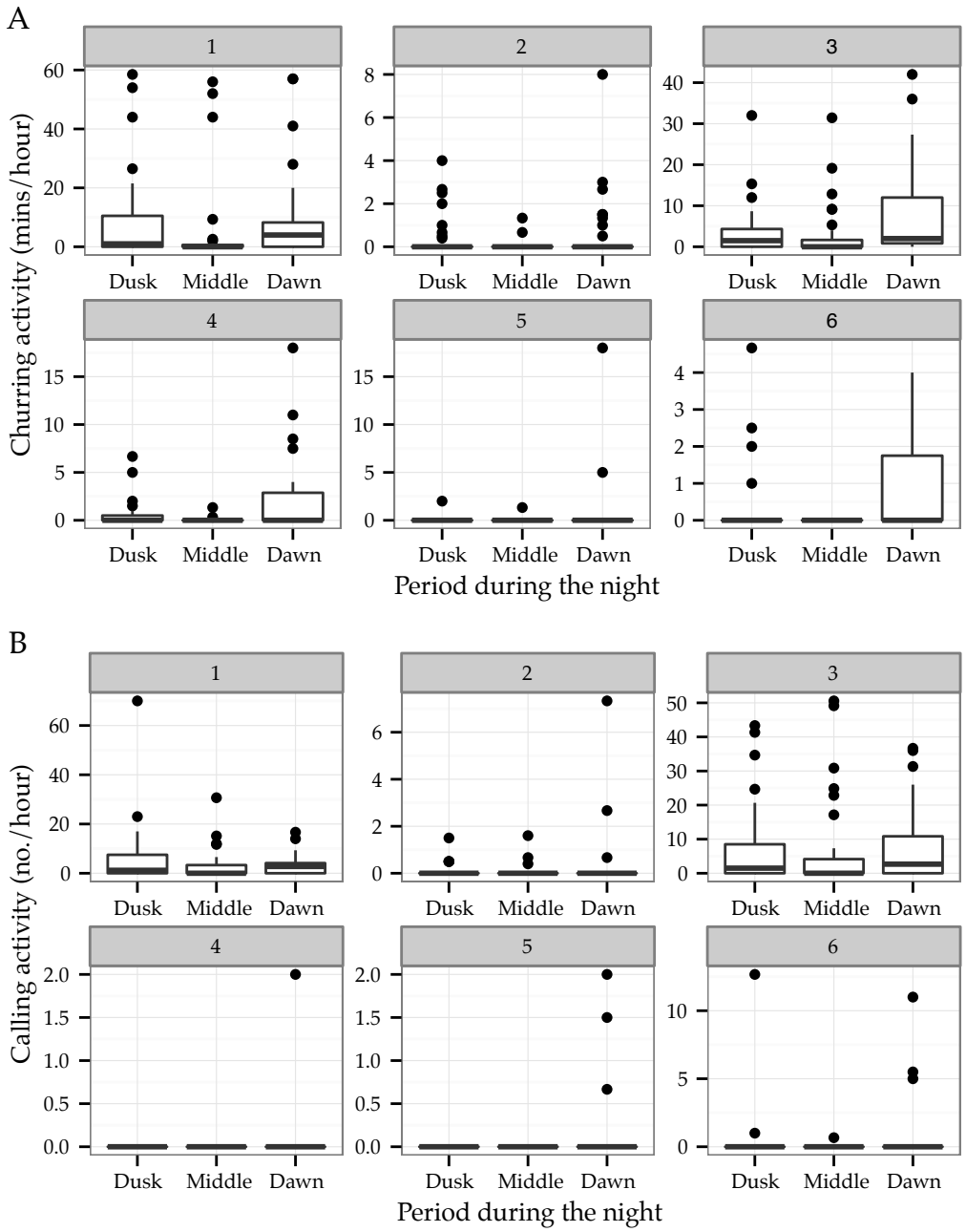

Figure 15: Nightjar activity at three different periods during the night as detected by automated recorders. These boxplots display (A) churring activity (measured in minutes of activity per hour) and (B) flight calling activity at six different automated recorders during the period of the survey: Dusk, 22.00-00.00; Middle, 00.00-02.30; Dawn, 02.30-04.30. The y-axes are distinct for each recorder because the amount of activity varied greatly according to location (and local abundance of nightjars). Most of the churring activity was recorded during the dawn period and the least amount of activity was during the middle of the night. There was no difference in flight calling activity between the three periods.

4.5.3 Comparison of recorder settings

We found that detection of nightjar activity varied among the different recorder settings. Having the recorder on only at dawn identified many of the nights that nightjars were active (75% in comparison to recording the whole night), while the recording time was substantially decreased (27% in comparison to recording the whole night). In addition, recording at both dawn and dusk also performed well. This setting detected almost all (98.8%) of the nights that there was some nightjar activity while only recording for 60% of the night. These results can be explained by our finding that nightjars are most active at dawn and at dusk. Setting the recorder to 10 mins every hour throughout the night had a varied performance; we tested two random sets of 10 mins every hour (e.g. 22.00-22.10, 23.00-23.10, etc and 22.30-22.40, 23.30-23.40, etc) and one performed significantly worse than the other ($p=0.036$). Recording only in the middle of the night (thus excluding dawn and dusk) had the worst performance of all the settings (see Table 7 and Figure 16).

4.6 Discussion

We compared the detection of nightjars from bioacoustic recorders deployed at the same time as experienced field surveyors, and found that recorders detected more nightjars (five out of eleven survey periods versus three out of eleven survey periods respectively). However, anyone wishing to detect nightjar presence with a recorder is unlikely to deploy the recorder for a few hours; instead, typical usage would involve leaving the device in situ for an extended period. When we compared the results from recorders deployed for a week, we found that bioacoustic recorders were significantly (and substantially) better at detecting nightjars than human surveyors. The recorders detected nightjars during 19 of 22 survey periods, while surveyors detected nightjars on only six of these occasions. In addition, there was no correlation between the amount of vocalisation detected by the recorders and the abundance of nightjars recorded by human surveyors – a correlation which would be expected if human surveyors detected most of the birds present. The recordings revealed that

Comparison	Estimate	Std Error	Z value	p-value adjusted
Dusk+Dawn – whole night	-0.016	0.081	-0.201	0.999
Middle – whole night	**-0.868**	**0.105**	**-8.268**	**<0.001**
Dusk – whole night	**-0.324**	**0.088**	**-3.698**	**0.004**
Dawn – whole night	-0.251	0.087	-2.877	0.060
10 mins – whole night	**-0.374**	**0.089**	**-4.209**	**<0.001**
10 mins2 – whole night	**-0.696**	**0.098**	**-7.083**	**<0.001**
Middle – Dusk+Dawn	**-0.852**	**0.105**	**-8.095**	**<0.001**
Dusk – Dusk+Dawn	**-0.307**	**0.088**	**-3.501**	**0.008**
Dawn – Dusk + Dawn	-0.235	0.087	-2.682	0.101
10 mins – Dusk+Dawn	**-0.358**	**0.089**	**-4.013**	**0.001**
10 mins2 – Dusk+Dawn	**-0.680**	**0.099**	**-6.899**	**<0.001**
Dusk – Middle	**0.545**	**0.111**	**4.919**	**<0.001**
Dawn – Middle	**0.617**	**0.110**	**5.591**	**<0.001**
10 mins – Middle	**0.494**	**0.112**	**4.425**	**<0.001**
10 mins2 - Middle	0.172	0.119	1.439	0.777
Dawn – Dusk	0.073	0.094	0.775	0.987
10 mins – Dusk	-0.050	0.095	-0.525	0.998
10 mins2 – Dusk	**-0.373**	**0.104**	**-3.570**	**0.007**
10 mins – Dawn	-0.123	0.095	-1.292	0.853
10mins2 – Dawn	**-0.446**	**0.104**	**-4.280**	**<0.001**
10 mins2 – 10 mins	**-0.323**	**0.105**	**-3.058**	**0.036**

Table 7: Comparison of nightjar detection by different recorder settings. The mean detection of nightjars by the different recorder settings is compared via Tukey Contrasts for the fitted Generalised Linear Model. The detection of nightjars is compared for the different recorder settings. Adjusted p-values are reported (single-step method) and significant p-values are given in bold. The different settings were: recording 10 mins every hour ("10mins" and "10mins2"), only dusk (22.00-00.00), only the middle of the night (00.00-02.30), only dawn (02.30-04.30), dusk and dawn ("Dusk+Dawn"), and recording all night ("whole night").

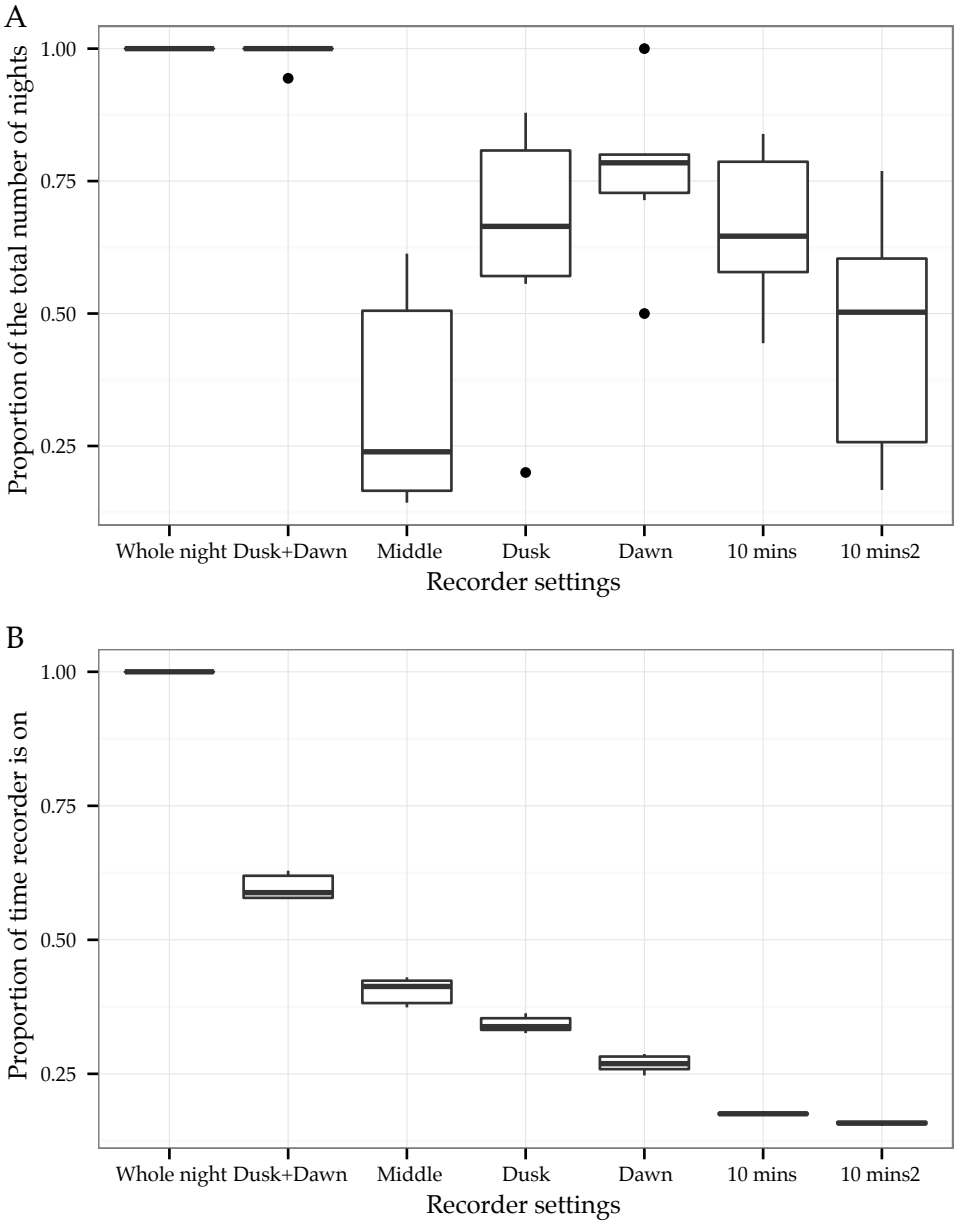

Figure 16: Proportion of the total number of nights nightjars were detected using different recorder settings. (A) We plotted the following recorder settings: dusk (22.00-00.00), middle of the night (00.00-02.30), dawn (02.30-04.30), dusk and dawn combined, and two random samples of ten minutes per hour ("10 mins" and "10 mins2"), and compared these to the total nights nightjars were detected when the recorders were left on during the whole night. (B) The relative length of each of these recording periods compared to recording the whole night. Our results indicate that most nightjar activity occurs during dusk and dawn; as a result, activating the recorders during these periods captures almost all (98.8%) of the nightjar activity, even though these periods make up only 60% of the night. The variability among the two 10-minute subsamples suggests that this is an unreliable sampling strategy for detecting nightjar activity.

nightjars were most active just before dawn and least active during the middle of the night. Recording at both dusk and dawn, or only at dawn, would give reasonably high levels of detection of nightjar activity while only recording for 60% and 27% of the night respectively.

While nightjars are not listed as one of the species that are difficult to survey in guidance by Joint Nature Conservation Committee (2004), our analyses reveal a major difference in detection by humans and recorders. This indicates that current survey methods for nightjars could be substantially improved, especially since we performed four surveys rather than the minimum of two surveys recommended by Gilbert et al. (1998). As nightjars are listed in Annex I of the Birds Directive, quantifying detectability for this species is important (Directive 2009/147/EC, 2010), with several Special Protection Areas (SPA) in the UK designated specifically for their breeding nightjar populations. SPAs in the UK are designated if an area is regularly used by >1% of the national population of a species that is listed on Annex I of the Birds Directive in any season. Our analyses indicate that bioacoustic recorders provide more accurate information on nightjar distributions, and could therefore be used to help better define SPAs.

One of the possible reasons for low detection by human surveyors could be that nightjars vocalise for short periods (e.g. we found vocalisations of <10 mins every hour (see Figure 15a)). Nightjars are a nocturnal species and thus surveys will be performed in low-light conditions, with surveyors relying mostly on their hearing to detect nightjars. The length of time that a human surveyor passes the location of a nightjar is probably of short duration minimising the probability of detecting a vocalising nightjar. As nightjars vocalise for short periods, increasing the frequency and/or length of stopping and listening for nightjars during a human survey may not significantly improve detection by human surveyors.

We found that nightjars are most active just before dawn or just after dusk. Therefore, our results confirm and extend the previous findings of Cadbury (1981), which were based on only a limited number of surveys. Our analysis showed that recorders could be set to record for shorter periods of time while achieving similar results, if knowledge of peak activity times is available. This

would save on battery life and data storage. As a result, the recorders could be left out for longer before the batteries would run out of power or before the data storage is full. For example, recording for two hours at dawn using SM2+ units without GPS would result in almost 13 nights of deployment (average battery life of 25.4 hours divided by two hours). Note that our average battery life of 25.4 hours results from our use of rechargeable batteries; if desired, recording time could be increased through the use of alkaline batteries or an external power supply such as a solar panel or a 6V or 12V battery source. We were not able to determine when nightjars are most active during the breeding season (e.g. during territory establishment or chick rearing). To answer this question, data from multiple years are needed as weather and other factors might have affected nightjar activity.

4.6.1 Advantages of using bioacoustic recorders

Our results suggest that bioacoustic recorders could be a very effective survey method for a variety of reasons. Firstly, surveys for species that do not vocalise regularly have a low accuracy as we have shown here in the case of nightjars. Secondly, surveying with bioacoustic recorders causes less disturbance than traditional surveys, as there is only the initial visit of deploying and picking up the recorders which could be done at a time when the species of interest is not active. Therefore, bioacoustic recorders could be used for species that are affected by human disturbance such as the capercaillie, short-eared owl and peregrine falcon (Joint Nature Conservation Committee, 2004). Thirdly, they can be of benefit when surveys are to be carried out in remote or difficult to access areas, as visits need only be made when deploying and picking up the recorders or replacing the batteries. Fourthly, there is no need for specialist surveyors to deploy or retrieve the recorders. Local people who know the area could deploy the recorders at a given location without the need for any detailed knowledge of the species of interest. Data could be analysed either by a specialist or, for species where a recognizer has been built (which will become more common as the use of automated recognition increases), by anyone with access to a personal computer. Currently, surveys for species that require a specialist include aquatic

warbler, capercaillie, and goshawk (Joint Nature Conservation Committee, 2004). However, it may be more difficult to use automated classification for the vocalisations of these species as these vocalisations might be more complex than those of the nightjar. In addition, there might be more background noise interference since these species are active during the daytime, but noise filters could be applied to the recordings to reduce this interference. A fifth advantage is that the cost of surveying using bioacoustic recorders is cheaper. We estimated the costs for an ecological consultancy company to carry out surveying work on one of the two sites from this study using field surveyors and automated bioacoustic recorders. A field survey approach would require four visits and would require 16 hours (based on two surveyors for Health and Safety reasons and assuming two hours per visit per person). In contrast, the approach using automated recorders would require three visits during the daytime: one visit to deploy the devices, one to replace the batteries and another visit to collect the devices. We estimate that this would require three hours (assuming one fieldworker). This needs to be offset by costs for processing the data from the recorders which we estimate to be two hours per device. On our sites we used three devices per site so assuming this, the processing time would total six hours. Thus, based on our theoretical example, a saving of seven hours work would be made using automated bioacoustic recorders, which saves 44% of time. Based on UK industry standards for an Ecologist or Senior Ecologist, this would be a saving of approximately £350 (based on £50/hour; source: Chartered Institute of Ecology and Environmental Management, http://www.cieem.net). Of course, the automated recorders would also provide a much richer source of survey data as they would be running for much longer periods than the field surveyors are present.

4.6.2 Further work on nightjars

This work has demonstrated the power of using bioacoustic recorders for determining the presence/absence of nightjar within a survey area. Traditionally, the presence of churring has been used as an indication of breeding activity and has been extrapolated to infer the numbers of breeding pairs present within an

area. However, churring per se is not an indication of pairing or breeding. It has been postulated (Andrew Lowe pers. comm.) that the call structure of the male nightjar is modified once a territory is established and mating is successful. If this hypothesis can be confirmed and the pre- and post-mating call can be quantified using bioacoustics, it may be possible to gain a more accurate measure of the breeding populations of this species, which to our knowledge has not been tried for this or any other species.

4.6.3 Conclusions

Here we have shown that bioacoustic recorders offer substantial improvements (217% increase in detection) over human surveyors in the detection of nightjars (an infrequently vocalising nocturnally active species) when deployed throughout the night. While previous studies have used bioacoustic recorders, they used interpreters to listen to the recorded audio files and classify the species on the recordings instead of automated classification. Listening to the audio files is very labour intensive, and automated classification could save a lot of time, especially when leaving the recorders out for longer periods (Swiston and Mennill, 2009). These previous studies detected similar numbers and species as observers conducting point counts in the field (Hobson et al., 2002; Rempel et al., 2005; Campbell and Francis, 2012) but only recorded for the duration of a traditional point count survey and did not investigate if recording for longer periods of time would increase performance, as is now possible with deployable recorders. Here we have shown that the accuracy of the recorders increases when they are deployed for a longer period of time. For example, when looking at the data recorded at the same time as the human surveys (22.00-00.00 on survey nights) nightjars were detected on five out of eleven instances (45%) by the recorders. However, the detection increased to eight out of eleven instances (73%) when we increased the selected recording time to the whole night of the survey (22.00-04.30 on survey nights; note that a 100% detection rate is likely unachievable as nightjars did not make regular use of some of the areas we surveyed). In addition, as we have performed line transect surveys, we also show for the first time that bioacoustic recorders can be used instead of line transect

surveys and not only to replace point counts.

Our study has implications for a range of other species, especially for species that are difficult to detect by traditional survey methods. For these species, we show that there is great potential to increase detection by using automated bioacoustic recorders. This is especially important when information on species presence and abundance is used to inform conservation.

4.7 Supporting information

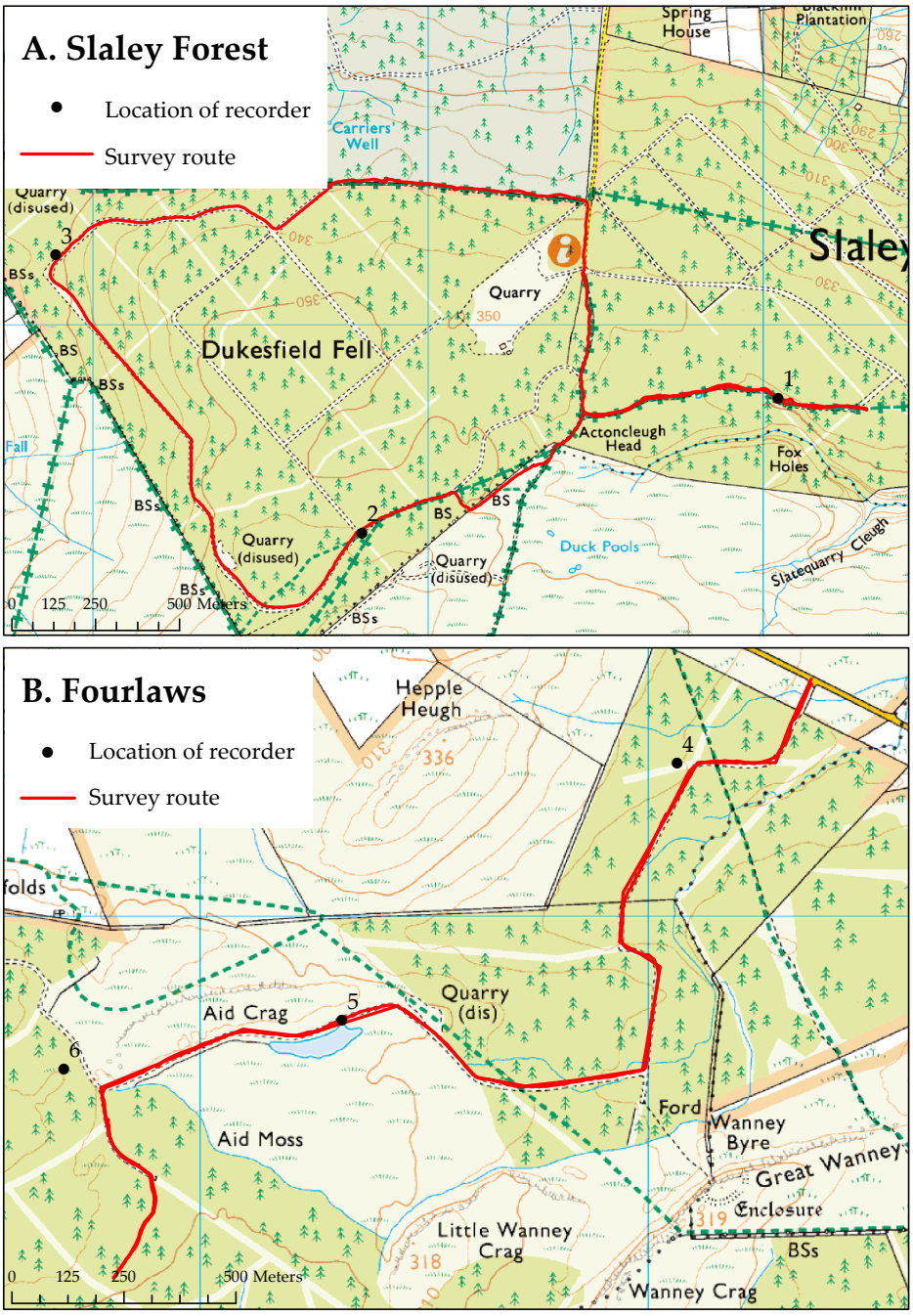

Figure S3: Map of study sites displaying the locations of the bioacoustic recorders and the survey route followed by human surveyors. Two Forestry Commission sites in Northumberland, UK, were surveyed: (A) Slaley Forest and (B) Fourlaws. Survey methods followed the methods described in Gilbert et al. (1998). At each site three full spectrum recorders (SM2+, Wildlife Acoustics Inc.) were placed, one per km^2 (six in total), during the same period as the traditional site surveys (between mid-June and the end of July 2012).

Time	Recorder 1	Recorder 2	Recorder 3	Recorder 4	Recorder 5	Recorder 6	Total
22:00-22:30	29	30	29	17	18	17	140
22:30-23:00	37	34	34	27	27	17	176
23:00-23:30	37	34	34	26	26	17	174
23:30-00:00	35	34	34	26	26	17	172
00:00-00:30	34	34	34	26	26	17	171
00:30-01:00	34	34	34	25	26	16	169
01:00-01:30	32	33	33	25	26	16	165
01:30-02:00	32	33	32	25	25	16	163
02:00-02:30	32	33	31	24	24	15	159
02:30-03:00	30	32	30	24	24	15	155
03:00-03:30	24	28	28	20	20	15	135
03:30-04:00	18	20	18	10	12	9	87
04:00-04:30	18	17	18	9	11	9	82

Table S1: Number of recordings for each 30-minute interval throughout the survey period. We deployed 6 SM2+ (Wildlife Acoustics Inc.) automated bioacoustic recorders (one per km^2) at two sites in Northumberland between mid-June and the end of July 2012. Recorders 1-3 were placed at Slaley Forest and recorders 4-6 at Fourlaws. The recorders were set to record throughout the night between 22.00 and 04.30 the following morning on both channels with a gain of +48dB and sampling rate of 44100Hz. This table shows the number of 30-minute recordings that were obtained for each recorder and time period; this number varies due to variation in battery use, which depends on local sound levels.

Sunrise	Sunset	Week	Recorder 1	Recorder 2	Recorder 3	Recorder 4	Recorder 5	Recorder 6
04:28	21:45	1	22:00-02:30*	22:00-03:00*	22:00-03:00*	22:30-03:00*	22:30-03:00*	
04:27	21:49	2	22:30-03:00	22:30-03:30	22:30-03:30	22:30-03:30	22:30-03:30	
04:32	21:48	3	22:30-03:00*	22:30-03:30	22:30-03:30	22:30-03:30*	22:30-03:30*	
04:38	21:44	4	22:00-03:30	22:00-03:30	22:00-03:30	22:00-03:30	22:00-03:30	22:00-03:30
04:48	21:36	5	22:00-04:30	22:00-04:30	22:00-04:30	22:00-04:30	22:00-04:30	22:00-04:30
04:57	21:27	6	22:00-04:30	22:00-04:30	22:00-04:30	22:00-04:30	22:00-04:30	22:00-04:30
05:09	21:16	7	22:00-04:30	22:00-04:30	22:00-04:30*	22:00-04:30*	22:00-04:30*	22:00-04:30
05:18	21:06	8	22:00-04:30	22:00-04:30	22:00-04:30			
05:27	20:56	9	22:00-04:30	22:00-04:30	22:00-04:30			

Table S2: Recorder sampling scheme over the survey period. We deployed 6 SM2+ (Wildlife Acoustics Inc.) automated bioacoustic recorders (one per km^2) at two sites in Northumberland between mid-June and the end of July 2012. Recorders 1-3 were placed at Slaley Forest and recorders 4-6 at Fourlaws. The recorders were set to record throughout the night between 22.00 and 04.30 the following morning on both channels with a gain of +48dB and sampling rate of 44100Hz. Each week the recorders were active until the batteries ran dry, which on average was four nights (range 3-5 nights). The traditional surveys took place at the beginning of week 1, 3, 5 and 7. An asterisk (*) denotes whether the recorders were on during the night of the traditional survey.

Setting	Configuration 1	Configuration 2	Configuration 3	Final configuration
Maximum complexity	32	42	16	16
Maximum resolution	6	8	11	15
Sample rate	16 kHz	6 kHz	8 kHz	8 kHz
FFT size	256	32	64	64
FFT overlap	1/2	1/2	1/2	1/2
Frequency minimum	9	6	8	8
Frequency maximum	49	10	20	18
Background filter	2s	1s	2s	1s
Maximum syllable length	48 ms	27 ms	100 ms	148 ms
Maximum syllable gap	48 ms	32 ms	100 ms	148 ms
Maximum song length	24 ms	60000 ms	10392 ms	5980 ms
Dynamic range	20	10	10	10
Algorithm	2.0	2.0	2.0	2.0

Table S3: Churring recognizer configuration settings. Table legend on next page.

Table S3: Churring recognizer configuration settings. We chose the initial recognizer configuration (Configuration 1) by manually experimenting with recognizer parameters in Song Scope until the true positives in our training dataset were correctly recognized. In Song Scope, when the log frequency scale with normalized power levels is selected, the program displays the recognized sections of the training data with hot colours. When recognizer settings are altered, Song Scope updates these recognized sections.

Configuration 2: FFT size and sample rate was reduced in order to get a better temporal resolution of the pulses that make up the churring of the nightjar. When FFT size and sample rate are adjusted the frequency minimum and maximum need to be adjusted as well in order to select the same frequency bandwidth. In addition, maximum song length was increased as the length of nightjar churring is long. Furthermore, we reduced the dynamic range in order to reduce the background interference.

Configuration 3: we reduced the maximum song length as alternative configuration 2 was giving too many false negatives. We also reduced the maximum complexity. Furthermore, we increased the maximum syllable length and gap. Alternative configuration 3 gave much better results (fewer false negatives and positives) than the previous two.

Final configuration settings: through further experimentation, we found that adjusting the maximum syllable length, maximum syllable gap and maximum song length provided additional improvements.

Discussion

Climate change affects humans through food security, water supply and human health (IPCC, 2007; Schmidhuber and Tubiello, 2007; IPCC, 2014). In addition, animals are also affected: for example, spring events are taking place earlier for a majority of species (Root et al., 2003; Parmesan and Yohe, 2003). As climate change is most likely to be caused by increases in GHG emissions (IPCC, 2012), there is an urgent need to reduce these emissions and renewable energy offers a potential solution. Wind power is an important source of renewable energy and has increased dramatically over the last decade (GWEC, 2013). While the exploitation of renewable energy sources will be fundamental to combating climate change, the rapid expansion of wind farm development has raised issues about potential harmful effects on wildlife, especially birds. Birds may be affected by wind farms directly through collision but also indirectly when the presence or operation of the turbines causes disturbance or displacement effects (Drewitt and Langston, 2006).

This thesis focused mainly on disturbance effects of wind farms and aimed to: (i) demonstrate the potential for use of existing bird monitoring data collected by EIAs and post-construction monitoring for developing an evidence base for the impacts of wind turbines on bird populations (Chapter 1); (ii) advance our understanding of the impacts of wind turbine noise on birds (Chapters 2 and 3); and (iii) to explore a potential new bird monitoring method using audio recorders (Chapter 4). Overall my thesis adds evidence to the impacts of wind farms on birds but also describes ways of using existing data and a potential alternative method of data collection. I outline these different areas below.

Using data from EIAs and post-construction monitoring to create an evidence base for wind farms

Potential effects of wind farms could be reduced substantially through careful placement of the wind farm and its turbines. Therefore, during the planning phase of a wind farm development, an EIA is carried out to inform of potential effects. When wildlife is judged to be at risk of harm planning applications from wind farm companies can be refused or subject to costly delays. These important decisions should be based on robust scientific evidence but due to the lack of data availability means that this is often not the case. Instead, planning decisions are typically based on collective experience or professional opinion (Hill and Arnold, 2012). However, the substantial amount of data generated from EIAs and post-construction monitoring could be used to inform planning decisions. Due to the specific purpose for which these data are gathered and the associated logistical issues of obtaining these data (despite being officially available in the public domain) means that they are rarely used to gain insights that could only be obtained from analysing data from a range of sites. For example, as I have outlined in my introduction, potential effects from wind farms are both species-specific and site-specific, and using data from EIAs and post-construction monitoring could be very valuable in understanding which species are affected and what the underlying mechanisms are that cause disturbance or collision. Therefore, I explored the usefulness of data which is routinely collected during EIAs and post-construction monitoring. I focused on black grouse, a species of conservation concern that is considered at risk from wind turbines (European Commission, 2011). I demonstrated in Chapter 1 that data from EIAs and post-construction monitoring could be used to answer important scientific questions and provide an evidence base for planning decisions.

Using data from seven wind farm sites, I showed that the abundance of lekking black grouse did not change in the areas around the wind farms but that black grouse leks generally within 500m of the nearest planned wind turbine moved locally after construction. I identified several reasons which, individually or in combination, could underlie these localised movements. These included the operation of the wind farm but also the increase in human activity on the sites,

changes in land management both within and surrounding the site, and habitat enhancement measures designed specifically for black grouse as mitigation for perceived impacts. Further study is needed to determine the causal factor(s).

Best practise

One limitation of EIA and post-construction monitoring data is that survey effort differs between years and between sites. This difference introduces variance and reduces the power to detect effects of interest. Best practise would be to have a greater consistency in survey effort which would make the data more useful. For example, the number of visits performed each year and the area to be surveyed should remain the same across the years. In addition, to reduce the difference in survey effort between sites, the survey boundary should be a particular distance from wind turbines for each site. For example, I recommended a survey boundary of 1.5km from wind turbines for black grouse in Chapter 1.

Data repository

A potential solution to the lack of data availability for the impacts of wind farms on birds is to create a central data repository, ideally populated by data from across many countries. Such a central repository would have data on the distribution of many species at different wind farm sites and this could aid research and planning decisions. It could help to increase our knowledge on which species in particular are affected by wind farms and what mechanisms are causing the effects. This in turn would significantly help planning decisions, as they would be more likely based on robust scientific evidence. In addition, during the planning phase mitigation could be put in place to reduce predicted negative impacts. The effectiveness of this mitigation could also be assessed via analysis of the data in the data repository. Recently, the Scottish Windfarm Bird Steering Group (www.swbsg.org, 2013) have started to create such a central repository for Scotland; hopefully similar efforts will eventually lead to an international data repository.

Impacts of wind turbine noise on birds

Wind turbine noise has been suggested as a potential reason which causes birds to avoid wind farm areas (Leddy et al., 1999; Drewitt and Langston, 2006; Hötker et al., 2006; Madders and Whitfield, 2006; Zeiler and Grünschachner-Berger, 2009). However, our current knowledge of the effects of wind turbine noise is limited and has been mostly extrapolated from the effects of urban and traffic noise. Urban and traffic noise have been shown to affect bird communication and have been linked to reduced densities and breeding success (Reijnen et al., 1996; Rheindt, 2003; Parris and Schneider, 2009; Halfwerk et al., 2011). As wind farms are built away from urban areas and therefore often free of other anthropogenic noise, it is especially important to assess whether similar negative effects can be expected for wind turbine noise and what these effects might be. My research has been a first step in advancing our knowledge of the effects of wind turbine noise on birds.

Predicting the effects of wind turbine noise

I investigated the potential effect of wind turbine noise on avian communication via a theoretical model. I simulated the propagation of an auditory signal in the presence and absence of wind turbine noise, and calculated the amount of reduction the sender of that signal would suffer after wind farm construction. Signals with frequencies below 2 kHz were predicted to be disproportionately masked by wind turbine noise and, therefore, species using these signals could have their communication substantially impeded. In addition, my model showed that other species vocalising above 2 kHz can also be affected but not as severely as species vocalising below 2 kHz. Furthermore, the reduction in area where the signal could be heard and interpreted (active space) was calculated for thirteen different species, some of them known to be affected by wind farms. The model predicted that the vocal communication of black grouse, wheatear and goldcrest were particularly affected by wind turbine noise and to a lesser extent the vocal communication of lapwing, golden plover, skylark, blackbird and willow warbler were predicted to be affected. If these species cannot adjust their vocalisations to overcome this masking effect of wind turbine noise, they might therefore avoid

wind farm areas. In addition, for black grouse, golden plover and wheatear, it is already known that they avoid wind farm areas, indicating that noise could play an important role in their avoidance of wind farms. In addition, in Chapter 1, I found that black grouse leks moved locally after construction. As their vocalisation, which is an important part in their lekking display, is substantially masked by wind turbine noise, this might have caused the movement of their leks.

Potential disturbance effects of wind farms underestimated

Whether wind farms have adverse effects on goldcrest, lapwing, skylark, blackbird and willow warbler is currently unknown. Further investigation is needed as to whether the masking effect of wind turbine noise on the communication of these species is causing population-level effects. Some of the population-level effects caused by wind farms might have gone unnoticed as the majority of studies assess disturbance by wind farms by investigating the distribution of birds at wind farm areas. For example, the number of breeding birds may remain high even though breeding success could be affected (Reijnen and Foppen, 1991; Habib et al., 2007; Francis et al., 2009; Slabbekoorn and Halfwerk, 2009) and therefore breeding numbers may give little insight into the mechanisms by which birds are affected (Halfwerk et al., 2011). In Chapter 2 I have shown that wind farm noise may affect the ability of birds to communicate with each other and given that other anthropogenic noise sources have been linked to reduced breeding success in birds (Habib et al., 2007; Francis et al., 2009; Halfwerk et al., 2011), breeding success in birds near wind farms could also be affected. It is thus important to focus on the effects of wind farms on specific elements of reproductive success to be able to understand whether and how birds are affected by wind farms.

Application to wind farm planning

There is considerable potential for the model to be used as a tool by policy makers and land managers. As this model can predict potential disturbance effects for specific species, it could be used as a tool during EIAs by identifying which species

will be affected by wind turbine noise. Especially for species for which there is no available literature on their disturbance effects of wind farms or for which the disturbance effects are in question, this model will be very valuable. The model can be easily adapted for the species and site in question.

In addition, the model could aid positioning of the turbines as assessment can be made at what distance the effect of wind turbine noise has negligible impacts on vulnerable bird species which may be present in the area. Furthermore, it could also help locating the wind farm itself by investigating which locations cause the least amount of communication disturbance for birds. For example, if locations that are investigated where the background noise levels differ substantially, the noisier location might be favoured as it might mask wind farm noise. Road noise has already been identified to mask wind turbine noise for humans (Pedersen et al., 2010) and therefore placing wind farms near roads could be advantageous. However, further studies are needed to investigate if road noise could mask turbine noise for birds. As well as the location of the wind farm, the model could also assess which type of wind turbine would cause the least amount of noise disturbance as the model can be adapted for any wind turbine type.

Wind turbine noise affects territorial defence

If we are to predict with accuracy the effects of anthropogenic noise on birds then a thorough understanding of the impacts on behaviour and life history are key. Birds depend on the communication of acoustic signals which are critical to survival. They need, for example, to be able to defend a territory, find a mate, beg for food and warn about the presence of a predator (Mathevon, 1997; Ten Cate et al., 2002; Riebel, 2003; Brumm, 2004b; Leonard and Horn, 2005; Leavesley and Magrath, 2005). However, the effects of noise on these signals are largely unknown. I investigated if and how wind turbine noise affects territory defence in European robins.

I found that European robins used low frequency elements to signal aggression and may therefore win territorial disputes while avoiding costly physical conflict. In contrast, in the presence of wind turbine noise, they reduced their usage of

low frequency elements when a territorial intrusion was simulated. They might have chosen signal detection over signal efficacy. As a consequence, it might reduce their ability to deter an intruder successfully which may result in the expenditure of extra time and energy and could also lead to an increase risk of injury. This in turn could lead to reduced reproductive success. As traffic noise also has most of its energy in the lower frequencies (similar to wind turbine noise), similar effects on territorial defence could be expected for traffic noise and these effects on territorial defence may be one of the causes of the reduced abundance and breeding success near roads. However, more targeted research is needed to confirm this. In addition, the consequences of reducing the usage of lower frequency elements need to be investigated. My research has been an important first step in advancing our understanding of the effects of noise on the defence behaviour in birds. However, birds also use other acoustic signals in order to survive and understanding the effects of noise on these other signals, such as begging calls, alarm calls and contact calls, is crucial to be able to predict the population-level effects of noise.

New bird monitoring method using audio recorders

Information on where species occur and in what numbers is important for conservation or as part of a legislative or policy process such as EIAs. A variety of sampling methods have been developed, including point counts, transect counts and mapping. However, these methods have several disadvantages, especially when the behaviour or ecology of the species of interest might substantially reduce their detectability. Automated recording via bioacoustic recorders might offer an alternative that might yield more detections and be more efficient. I investigated this surveying method in comparison with traditional human surveys using the European nightjar as an example.

I found that bioacoustic recorders were substantially better at detecting nightjars than human surveyors. This indicates that survey methods for nightjars could be significantly improved by using bioacoustic recorders. As nightjars are listed in Annex I of the Birds Directive, several SPAs are specifically designated for their

breeding nightjar populations. This new surveying method could be used to help delineate SPAs more accurately.

Bioacoustic recorders could be used for any species that vocalise loudly enough to be recorded at reasonable distances and will probably substantially improve detection for species where their behaviour or ecology reduces detectability during traditional human surveys. In addition, these bioacoustic recorders can be used for species that are affected by human disturbance because the initial visit when recorders are deployed and the final one when they are picked up, could be undertaken at a time when the species of interest is not active. Other advantages include: there is no need for a specialist surveyor to deploy or pick up the devices, reduced costs and increased survey effort in remote areas.

I used automated classification to identify nightjar vocalisations on the recordings. Nightjars have relatively simple vocalisations and are only active at night, and therefore its vocalisation lends itself to automated classification. For other species with more complex vocalisations and/or are active during the day when background noise levels may be higher, it might be more difficult to use automated classification. Further research is needed to assess for which species current automated classification methods can be used. For the species to which current methods cannot be applied, new automated classification methods need to be developed.

Because bioacoustic recorders will give more accurate information on species abundance and their distribution, a more accurate prediction of the effects of the developments can be made. This will aid EIAs and post-construction monitoring for wind farms and other developments. For example, the recorders could aid in assessing the effects of wind farms on wildlife as they could be used to assess species activity before and after the construction. Changes in activity such as timing of the dawn chorus could be investigated. In addition, there will be potential to investigate noise effects on wildlife by analysing the songs or calls on the recordings of the species of interest, including bats, birds and mammals. As these recorders could also be deployed in marine locations for long periods of time, they could be very valuable in determining the effects of off-shore wind

farms on marine wildlife.

Conclusion

It is important to consider the biodiversity of the proposed development sites if the conversion from electricity generation based on fossil fuels to that of wind is to be successful. Crucially, biodiversity gains made by lessening the negative effects of climate change must not be offset by biodiversity losses incurred through the incorrect placement and operation of wind farms. It is therefore important to understand the effects of wind farm development on wildlife. In this thesis, I have demonstrated that routinely collected data from EIAs and post-construction monitoring can be used to develop an evidence base for the impacts of wind farms on bird populations. A central repository could be created to deal with the lack of data availability and will aid to improve our understanding of the potential effects that wind farms could have on birds and what mechanisms drive these effects. Especially for species that are of conservation concern and where the effects are currently unknown or unclear, this data repository could be very valuable. For example, the data repository could be important for species that use open landscapes such as black grouse, wheatear, golden plover, lapwing and skylark. Especially since the acoustic communication of these species are predicted to be substantially affected by wind farm noise as shown in Chapter 2.

Assessing the quality of proposed wind farm sites could be improved by using the model I have developed in Chapter 2. The model predicts which species are affected by wind turbine noise and could aid the placement of wind farms and their turbines. In both Chapter 2 and 3, I have shown that noise could play an important role in the disturbance effects of wind farms on birds. In addition, as the model can predict which species are likely to be most at risk from noise effects, this can guide which species should be of particular concern for data collection to populate the data repository. For example, the effects of wind turbine noise on species that vocalise below 2 kHz are predicted to be significant. Therefore, the data repository could be important to identify the disturbance effects of wind farms on species that vocalise in this range. However, common

species like blackbird, goldcrest and willow warbler should not be overlooked as the active space of these species is predicted to be seriously reduced.

A thorough understanding of the impacts on all aspects of behaviour is important to accurately predict the effect of wind farms on birds. Successful communication of acoustic signals which are essential to survival is crucial and birds rely on them in order to survive and reproduce. I have shown that territorial defence behaviour in robins is affected by wind farm noise. Further investigation is needed to see if and how defence behaviour of other birds is affected. Although my research has been an important first step in advancing our knowledge of the effects of noise of bird behaviour, birds also use other acoustic signals in order to survive and understanding the effects of noise on these other signals, such as begging calls, alarm calls and contact calls, is crucial to be able to predict the population-level effects of wind farm noise and thus also of wind farms.

It is important to understand the effects of wind farms on wildlife in order to successfully switch from fossil fuels to wind electricity generation. And to be able to understand these effects, data on species distribution and abundance are fundamental. Various species have reduced detectability during traditional surveys due to their ecology or behaviour. Therefore, understanding the effects of wind farms on these species will be challenging. However, I have demonstrated that surveying using bioacoustic recorders could significantly increase detection of these species and thus give more accurate information on where they live and when they use the site. Therefore, monitoring with bioacoustic recorders could offer a potential solution and could aid in assessing the effects of wind farms on these species as well as others. In addition, this method could be used during EIAs and post-construction monitoring.

While we might not be able to prevent all negative effects of wind farms on wildlife, understanding these effects can substantially reduce them. A thorough understanding of the effects of wind farm development on wildlife will be fundamental in advising planning decisions and wind farm placement. Most adverse effects on birds can probably be prevented by appropriate placement of wind farms, and EIAs provide vital means to do so. When negative impacts are

either predicted or measured for a range of factors (including wildlife), mitigation measures can be put in place in order to reduce these effects. And once done, we might be able to successfully switch from electricity generation based on fossil fuels to renewable energy and reduce GHG emissions sufficiently in order to halt global warming.

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